Assessment of the Aquatic Effects of Mining in Canada:

AQUAMIN

Supporting Document II: Appalachian Region

Case Studies

103-05106/

Assessment of the Aquatic Effects of Mining in Canada: AQUAMIN

Supporting Document II: Appalachian Region

Case Studies

Prepared for:
AQUAMIN Steering Group

By:
AQUAMIN Working Group 3

1996

Working Document

Contents

Page

1 East Kemptville Tin Mine 1
1.1 General Site Description 1
1.1.1 History and Background 1
1.1.2 Acid Precipitation 5
1.2 Assessment of the Monitoring Programs 6
1.2.1 Water Quality 6
1.2.2 Sediment Quality 9
1.2.3 Benthos 10
1.2.4 Fish - Population Assessment 11
1.2.5 Fish - Tissue Residues 12
1.2.6 Toxicity Testing 13
1.2.7 Zooplankton 15
1.3 Results of Aquatic Effects Assessment 15
1.3.1 Water Quality 15
1.3.2 Sediment Quality 16
1.3.3 Benthos 17
1.3.4 Fish - Population Assessment 17
1.3.5 Fish - Tissue Residues 18
1.3.6 Toxicity Testing 19
1.3.7 Zooplankton 19
1.4 Discussion 19
1.5 Conclusions 21
1.6 Recommendations 22
1.7 References 23

2 Heath Steele Mines 24
2.1 General Mine Site Description 24
2.1.1 Geological Setting 24
2.1.2 History and Type of Operations 29
2.1.3 Site Modifications and Efforts to Reduce Acid Mine Drainage 30
2.1.3.1 Heath Steel Mine/Mill Site 30
2.1.3.2 Stratmat Mine Site 31
2.1.4 Wastewater Disposal 32
2.2 Quality and Quantity of Information for Mine Site 33
2.2.1 Description of Monitoring Programs 33

2.2.2 EEM Methods and Tools 36
2.2.2.1 Water Quality 36
2.2.2.2 Sediment Quality 37
2.2.2.3 Benthos 37
2.2.2.4 Toxicity 38
2.2.2.5 Fish Populations 38
2.2.2.6 Fish Tissue Analysis 38
2.2.2.7 Adequacy of Monitoring 39
2.3 Magnitude and Extent of Environmental Effects 42
2.4 Costs and Benefits of Monitoring Program 49
2.5 Compliance Record in Relation to Effects 49
2.6 Discussion 51
2.6.1 Relationship Between Physical and Biological Effects 51
2.6.2 Do the MMLEG Adequately Protect the Environment? 54
2.6.3 Can the EEM at Heath Steele Mines be Improved? 55
2.7 Conclusions and Recommendations 56
2.8 References 61

3 Hope Brook Gold Mine 66
3.1 Introduction 66
3.1.1 Site Description 66
3.1.2 History and Type of Operations 66
3.2 Assessment of Monitoring Programs 69
3.2.1 Water Quality 70
3.2.2 Sediment Quality 72
3.2.3 Fish - Tissue Residues 74
3.2.4 Invertebrates - Tissues Residues 74
3.2.5 Invertebrates - Population Assessment 77
3.3 Results of Aquatic Effects Assessment 77
3.3.1 Water Quality 77
3.3.2 Sediment Quality 78
3.3.3 Fish - Tissue Residues 78
3.3.4 Invertebrates - Tissue Residues 79
3.3.5 Invertebrates - Population Assessment 79
3.4 Discussion 79
3.5 Conclusions 80
3.6 Recommendations 80
3.7 References 80

1 East Kemptville Tin Mine

1.1 General Site Description

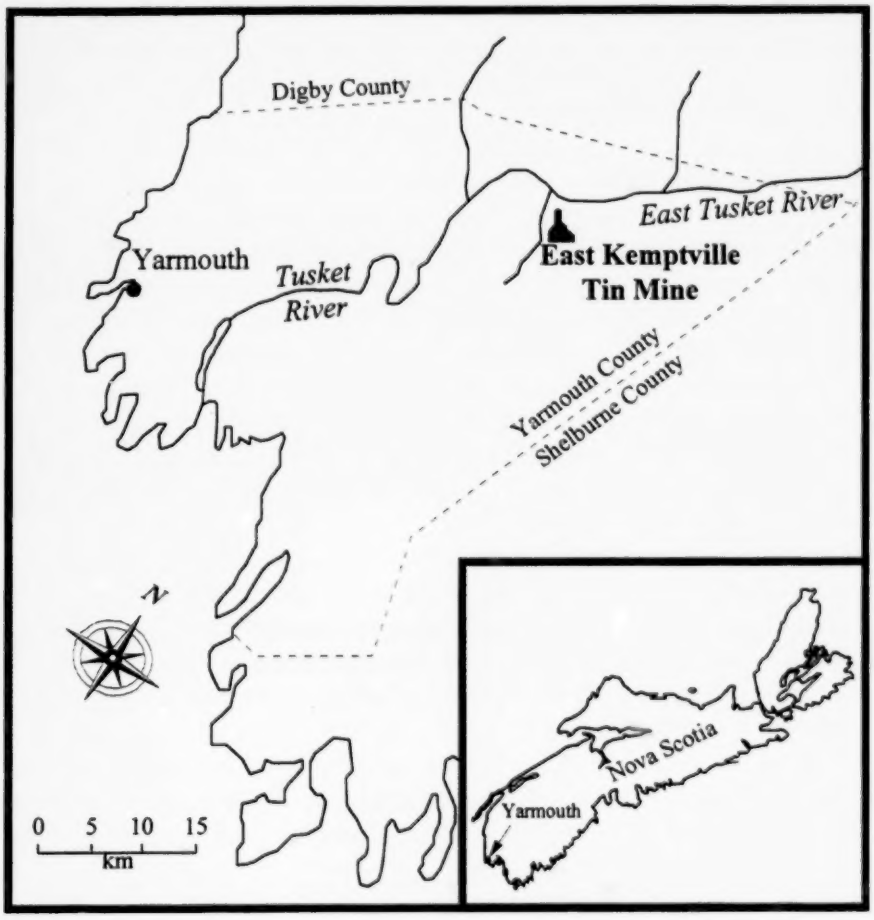
1.1.1 History and Background

The East Kemptville Tin Mine is located in Yarmouth County, Nova Scotia, approximately 45 kilometres northeast of the Town of Yarmouth (Figure 1.1). The deposit was first explored by Shell Resources Canada in the mid-1970's. In 1981 Shell Resources decided to concentrate on oil and natural gas projects and to cease activities in mineral production. The property was sold to Rio Algom Ltd. in 1982. Rio Algom continued with previous plans made by Shell and began construction of the mine in 1984. Construction was completed and production started in October, 1985. The mine operated until reduced ore grades and low metal prices forced its closure in January, 1992.

The mine was the only primary tin producer in North America and it was the largest open pit tin mine in the world. The tin was found in a weathered granite host rock which also contained a small amount of a silica-fluoride complex. The ore was extracted from an open pit mine at a rate of 9000 tonnes per day. The ore was reduced in size to less than 18 cm in a primary crusher before being fed into the mill. Through further crushing and grinding, the ore was reduced to a particle size of less than 1 mm. Recovery of the tin was accomplished by gravity separation and flotation. The tin was separated from the granite host rock through a series of Reichert cones, spiral circuits and shaker tables. Tin was also recovered from the finer ore particles by means of froth flotation. Copper and zinc concentrates were recovered by means of additional flotation circuits. On average, the mine produced about 35 tonnes of tin concentrate, 15 tonnes of copper concentrate and 15 tonnes of zinc concentrate per day. Tin concentrates were packed loose in 20 tonne shipping containers which were transported by truck to Halifax for shipment to smelters and refineries via container ship. Copper and zinc concentrates were shipped via truck and rail to smelters/refineries in central Canada.

Three solid wastes were generated by the mill circuit: coarse, slime and float tailings. These waste products were discharged via two separate slurry pipelines to an on-site, engineered tailings basin. The tailings were deposited so that the fine tailings (slime and float), which contained most of the original sulphide minerals, were placed on the bottom of the pond and were covered with the coarse tailings. The coarse tailings were primarily a coarse silica sand. Due to the presence of sulphide minerals in the fine tailings, this waste product could potentially produce acid mine drainage if allowed to oxidize. By placing them under water and covering them with the coarse tailings, it was felt that the formation of acid mine

Figure 1.1: Location of the East Kemptville Tin Mine



drainage could be minimized.

The tailings pond covered about 304 hectares and was originally divided into two cells. The large first cell was used for the deposition of the tailings. The second cell, 27 hectares in area, was used for final settling of particulates and as a reclaim water supply. During the winter and spring of 1986, this arrangement proved ineffective at removing fine particulate material from the effluent resulting in elevated levels of suspended solids in the effluent and in the receiving waters. In order to correct this problem, the second pond was divided into 3 sections: a reclaim pond, a settling pond and a polishing pond. A water treatment plant was built between the reclaim pond and the settling pond. Water from the reclaim pond was

recycled to the concentrator for use as process water. This practice reduced the volume of fresh water required in the process. Excess water from the reclaim pond was treated with the addition of a polymer-based flocculent to promote the precipitation of the fine particulates. The water treatment plant also added lime to raise the pH of the effluent to promote the precipitation of metals.

Effluent from the site was discharged to Duck Pond Brook which flows into Big Meadow Brook, a tributary of the East Tusket River (Figure 1.2). Effluent samples taken by the company and the regulatory agencies indicate that the effluent was generally in compliance with the *Metal Mining Liquid Effluent Regulations* (MMLER)(Table 1.1). These water samples were collected between September 19, 1984 and October 26, 1994. The data indicates that the mine was out of compliance with MMLER's for 6 of 14 data sets in 1987. In subsequent years non-compliance occurred in only one month until mine closure in January of 1992. During the period from 1988 until 1991, total zinc analysis in the effluent was above the MMLER limit of 1.0 mg/L for a grab sample 10 times for 151 samples. The maximum zinc concentration was 2.07 mg/L. Total suspended solids in the effluent was above the MMLER limit of 50 mg/L for a grab sample 4 times for 146 samples.

Site specific water quality objectives were established for the East Tusket River for fourteen parameters (Table 1.2). A monitoring program with sampling and analytical protocols was designed to monitor the effectiveness of these objectives as a management tool. The assessment point for the water quality objectives was below the confluence of the East Tusket River and its tributary Big Meadow Brook. Background reference stations were designated one for each watercourse upstream of the mine's influence. Pre-operational baseline data was available for each station. A monitoring committee was established in 1987 to review the results of the monitoring program and to determine if the water quality objectives were being met. The committee prepares an annual report on the results of the water quality monitoring program and their review of those results.

Figure 1.2: East Tusket River drainage, and effluent discharge from the East Kemptville Tin Mine

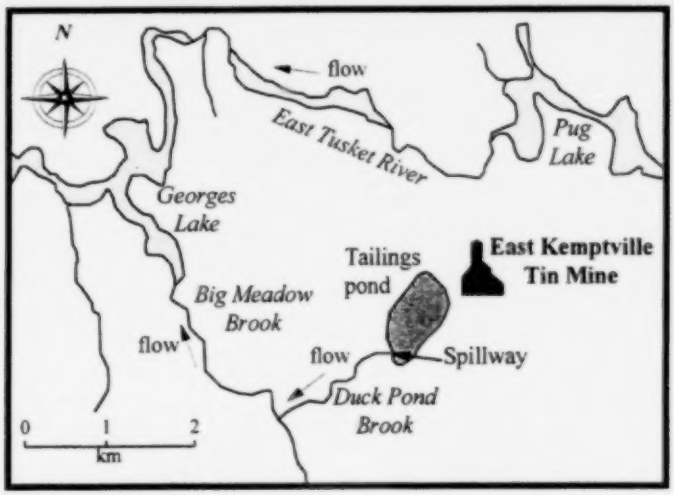


Table 1.1: Effluent Limits Industrial Waste Permit

Effluent limits established for the East Kemptville Tin Mine by the approval of the Industrial Waste Treatment Works by the Nova Scotia Department of the Environment on February 16, 1984.

Parameter	Maximum in a grab sample	Maximum in a composite sample	Monthly Mean
pH (1,2)	5.0	5.5	6.0
		mg/L	
Iron	7.0	-	-
Suspended solids (1)	50	37.5	25
Copper (1)	0.6	0.45	0.3
Zinc (1)	1.0	0.75	0.5
Arsenic (1)	1.0	0.75	0.5
Lead (1)	0.4	0.3	0.2
Nickel (1)	1.0	0.75	0.5

(1) From *Metal Mining Liquid Effluent Regulations*
(2) pH values shown are minimum values, not maximum

Table 1.2: Water quality objectives for the East Tusket River

Parameter	Objective	Comparison Criteria
pH	4.3	25th Percentile
Suspended Solids	25 mg/L	Median
Cyanide	0.005 mg/L	Median
Fluoride	1.0 mg/L	Median
Iron	0.3 mg/L	Median
Zinc	0.03 mg/L	Median
Arsenic	0.01 mg/L	Median
Nickel	0.5 mg/L	Median
Manganese	0.04 mg/L	Median
Aluminum	0.5 mg/L	75th Percentile
Cadmium	0.005 mg/L	75th Percentile
Copper	< 0.01 mg/L	75th Percentile
Lead	0.005 mg/L	75th Percentile

1.1.2 Acid Precipitation

The waters of the East Tusket River have been adversely affected by the long range transportation of air pollutants from central Canada and the northeastern United States. The pH of precipitation in south-west Nova Scotia averaged 4.61 between 1978 and 1987 (Underwood, *et al.* 1989). Since the waters of the East Tusket River are poorly buffered (hardness = 3 mg/L; alkalinity = 1 mg/L), the acidic precipitation has caused a depression in the pH of the river water. The average pH of the water in the East Tusket River upstream of the mine was 4.6 between 1984 and 1994 and in summer, during low flow conditions, levels as low as 3.7 have been observed. Fish species which are sensitive to acidic conditions, such as the Atlantic salmon, are no longer able to complete their life cycle in the East Tusket River and some of its tributaries.

1.2 Assessment of the Monitoring Programs

1.2.1 Water Quality

AQUAMIN Sources:

Report Nos. 621 and 713 were prepared as background for a pre-operational Environmental Impact assessment of the proposed mine site. The data included base line information useful for any future monitoring.

Report Nos. 346, 629, and 668 were carried out after the mine became operational. They were conducted to assess sediment contaminants, and benthic and fish communities. The water sampling and analytic data contained in study #346 provides an in depth assessment of the mine's impact on receiving waters. This study contains water quality data from background and downstream monitoring stations and therefore provides excellent source information for AQUAMIN.

In addition, Report No. 346 documents sampling protocols as well as field and laboratory analytic methodologies. Quality control techniques were used for both field and laboratory.

The information contained in the AQUAMIN Database Reports, when interpreted with other available monitoring data, provides the necessary documentation to conclude that the water quality monitoring is adequate to protect the receiving environment.

Sources External to AQUAMIN Database:

In 1984 Rio Algom established a baseline surface water monitoring program for the waters near the mine site. Shortly after the mine became operational, concerns over elevated levels of suspended solids in the effluent discharge led to the establishment of site specific water quality objectives for the East Tusket River. Objectives were established for fourteen parameters. These objectives were based on environmental effects of the selected chemicals to aquatic organisms and were formulated after evaluation on a one-by-one basis.

Water sampling conducted by Rio Algom consisted of monthly grab samples for over 30 parameters at eight locations. The samples were preserved on site and transported to the laboratory at the Victoria General Hospital in Halifax for analysis. The results of the analyses were compiled and computerized by Rio Algom. The data was submitted to Conservation and Protection Branch of Environment Canada and the Nova Scotia Department of the Environment. Both of these regulatory agencies conducted their own surface water quality monitoring programs.

The NSDOE has carried out monthly grab sampling at four of the sampling locations also utilized by Rio Algom. In addition, Environment Canada conducted surface water sampling a minimum of twice a year. When possible, this sampling was carried out at the same time as sampling by Rio Algom. These samples were analysed independently and the results compared as part of the quality control program.

The water quality data from these sources was compiled using a Lotus 1-2-3 spreadsheet. Means, minimum and maximum values, medians, and percentiles were calculated for key parameters including those for which water quality objectives had been established (Table 1.3).

A pre-operational environmental impact assessment has provided baseline data for 25 parameters at 9 stations. Water quality objectives were established for the East Tusket River for 14 parameters. The assessment point at the confluence of East Tusket River and Big Meadow Brook is five kilometres downstream of the tailings pond discharge. Sequential triplicate sampling was carried out. One way ANOVA was used to analyse samples for fluoride levels in receiving waters.

Annual water quality summaries have been prepared by the Conservation and Protection Branch of Environment Canada and submitted to the East Tusket River Water Quality Monitoring Committee for review and recommendation to the Regional Director General.

With the exception of the initial suspended solids in the effluent and an occasional elevated level for zinc, this mine has operated in compliance with MMLER's and the Industrial Waste Permit issued by the NSDOE. The site specific water quality objectives have not been exceeded and have served as an effective monitoring tool.

The Summary of Surface Water Quality Data from 1984 to 1993 indicates that the receiving waters were not negatively impacted as a result of the Mine's effluent discharge. The water quality immediately downstream of the mine appears to have been enhanced due to the increase in pH as a result of the lime treatment of the effluent sent to discharge.

In conclusion, it is evident that the water quality monitoring and analytic protocols for surface water at this mine are more than adequate to identify and quantify any aquatic effects on the receiving waters.

Table 1.3: Summary of water quality monitoring at the East Kemptville Tin Mine: 1986 to 1994

	Al	As	Cd	Cu	Pb	Mn	Zn	Fluoride	Hardness	TSS	pH
						mg/L					
FINAL EFFLUENT											
Number	136	141	124	150	149	151	151	140	137	146	144
Mean	0.83	0.009	0.006	0.02	0.003	0.43	0.37	9.81	257.99	11.55	7.47
Minimum	0.05	0.002	0.002	0.01	0.002	0.01	0.02	0.10	5.10	0.30	5.30
Maximum	6.40	0.150	0.027	0.30	0.050	1.73	2.10	19.00	569.00	566.00	9.70
Standard Dev.	0.83	0.016	0.004	0.03	0.005	0.34	0.37	4.74	185.64	50.22	0.87
Mediam	0.585	0.005	0.005	0.01	0.002	0.32	0.23	11.1	234.9	2.8	7.37
75th Percentile	1.03	0.01	0.01	0.02	0.0	0.61	0.50	13.13	448.00	7.18	8.00
FUG LAKE (UPSTREAM)											
Number	126	127	121	132	130	132	132	125	127	129	129
Mean	0.36	0.004	0.00	0.01	0.002	0.02	0.02	0.11	2.96	2.03	4.64
Standard Dev.	0.17	0.001	0.00	0.02	0.001	0.02	0.03	0.08	0.96	3.37	0.49
Maximum	1.76	0.005	0.01	0.26	0.007	0.20	0.28	1.00	6.05	29.30	7.40
Minimum	0.02	0.002	0.00	0.00	0.001	0.01	0.01	0.10	1.23	0.30	4.10
Median	0.34	0.01	0.00	0.01	0.00	0.01	0.01	0.10	2.85	1.30	4.50
75th Percentile	0.43	0.01	0.00	0.01	0.00	0.02	0.01	0.10	3.40	2.00	4.70
ASSESSMENT POINT (DOWNSTREAM)											
Number	118	117	116	119	118	119	119	116	117	116	116
Mean	0.37	0.004	0.002	0.01	0.002	0.047	0.03	0.63	15.17	1.43	4.86
Standard Dev.	0.11	0.002	0.001	0.00	0.001	0.183	0.03	0.75	22.48	1.00	0.47
Maximum	0.66	0.020	0.010	0.04	0.013	2.010	0.19	5.80	196.0	6.50	6.50
Minimum	0.18	0.002	0.001	0.00	0.002	0.010	0.01	0.10	2.60	0.04	4.20
Median	0.35	0.01	0.00	0.01	0.00	0.02	0.01	0.40	9.04	1.20	4.77
75th Percentile	0.46	0.01	0.00	0.01	0.00	0.03	0.03	0.63	16.80	2.00	5.03

1.2.2 Sediment Quality

Two pre-operational sediment sampling campaigns were completed for this site. There was no indication of any quality control or quality assurance for either report. Descriptions of sampling methodology and analytical techniques were non existent. No attempt was made to relate the sediment assays to aquatic effects.

Report No. 713:
A total of nine locations were sampled. There was no reference on Big Meadow Brook upstream of Duck Pond Brook. The following were analysed for each station: moisture, ash, organic, pH, zinc, copper, lead, tin, tungsten, arsenic, mercury and grain size analysis. All values for metals in sediments were low with the exception of arsenic in sediments in both Duck Pond Brook and Big meadow Brook being 2 to 3 times above background. The zinc levels in the sediments in Duck Pond Brook were about 2 times background.

Report No. 621:
A total of 12 samples were analysed for arsenic only. The arsenic levels were comparable to background levels, including Duck Pond and Big Meadow Brooks. No explanation was offered for the anomaly of high arsenic readings in the previous year's study (Report No. 713).

The data contained in both of these pre operational reports must be used with some caution as there was no QA/QC nor sampling and analytical methodologies described. The sediment quality before the mine began operation was not seen to have a significant effect on the aquatic environment.

Report No. 629:
One sediment sampling campaign was completed when the mine was in operation. This study was completed in 1986 after a period where the effluent from the tailings pond contained elevated suspended solids.

A total of 6 samples were taken and analysed for 22 metals, total organic carbon and total organic nitrogen. Cores were also taken at each site for visual inspection of the sediment. Reference stations were incorporated for both the East Tusket River and Big Meadow Brook. There was no reference station on Duck Pond Brook as the brook is now part of the tailings pond. The quality control was very well documented with blind triplicates taken at each location. No analysis has been completed on the QC data, however. Sampling and analytical procedures are well documented and are more than adequate. All samples were submitted

blind to the labs.

Examination of the sediment cores did not indicate the presence of the gray coloured kaolinite typical of the tin mine tailings. Also the grain size analysis indicated that there was no influence of the mine operation on the grain size distribution of the sediments. Metal levels in the sediments show some interesting trends. The chromium levels in all the samples was high. Levels in the Tusket River and Big Meadow Brook reference sites were 2 to 3 times higher than in Duck Pond Brook. Arsenic levels in all sediments were also higher in the reference stations than the downstream stations. Zinc levels in the sediments was 3 to 4 times above background in locations downstream of the mine. The other metals were slightly elevated in the sediments of Duck Pond Brook while they were at background levels in the East Tusket River and Big Meadow Brook. The sediment quality of the East Tusket River and Big Meadow Brook was unaffected by the operation of the mine.

Report No. 629 was the only report with adequate, reliable sediment quality data. The data in the report indicates that the only detectable impact on sediment quality due to the mine was in Duck Pond Brook. The other rivers in the area were largely unaffected.

1.2.3 Benthos

Only two of the reports contained data on benthos that was reliable enough to draw conclusions from, one pre-operational (Report No. 713) and one during operation (Report No. 668). Report No. 668 was preliminary in nature.

Report No. 713:
This report describes the pre operational survey done at the site. A total of 7 locations sampled, 6 with Surber and 1 with Ekman. Surbers were done in triplicate, Ekman as a single. Procedures well documented. Reference samples were included for the Duck Pond Brook, Big Meadow Brook and the East Tusket River. Results were consistent for each site with diversity indices of about 2 and biotic indices of 1.6. Results indicate a healthy benthic community for unproductive northern waters.

Report No. 668:
This report summarizes a preliminary investigation completed in response to the high suspended solids noted in the tailings pond effluent from the site in 1985-1986. A total of 8 samples were taken 4 Surbers, 4 Ekmans. No QA/QC. The benthic community in Duck Pond Brook was impacted compared to pre-operational survey. There was evidence of some pond/lake type of organisms in Duck Pond Brook. The explanation was that they probably

came from the tailings pond which was filled with water from Pug Lake. All other rivers were comparable to baseline study. Stress on Duck Pond Brook may be due to the fact that the brook was dry for an extended period while the tailings pond was filling.

Report No. 629:
This report describes a detailed follow up investigation to Report No. 668. Sampling included 6 stations in riffles for Surber samples and 6 locations in slow moving pools for Ekman dredge samples. The pools were sampled because these are the places where the solids from the tailings dam effluent are most likely to deposit. All samples were triplicate and methods were well described. Reference stations were sampled on Big Meadow Brook and the East Tusket River. A control sample was taken on Bear Lakes Brook. Duck Pond Brook had fewer taxa and organisms than the in the baseline survey. All other sample locations were comparable to pre-operational conditions with evenness values above 0.6 and diversities greater than 3.0. The samples from the pools above and below the influence of the mine effluent did not produce enough organisms to permit interpretation of results. No explanation is offered in the report. The report concludes that Duck Pond Brook was the only stream affected by the construction\operation of the mine.

The data quality for the benthic sampling was adequate. The two major studies included triplicate samples and reference samples. No analysis of the sample variance was done. At this early stage of the mine's operating life the impact of the effluent on the benthic community in the receiving environment was confined to Duck Pond Brook. Benthos in Big Meadow Brook and the East Tusket River were unaffected as compared to the pre operational studies. One report hypothesizes that the effects in Duck Pond Brook may be the result of stress on the benthic community from the period when the tailings pond was filling and the brook was basically dry. There is no data to support this statement. In order to quantify longer term effects of the mine effluent on the benthic invertebrate community in the receiving waters, more current sampling information is required.

1.2.4 Fish - Population Assessment

One study (Report No. 629) looked at the fish populations during 1986, shortly after the mine commenced operation. The study was rather ad hoc in nature but was most useful in qualitatively describing species captured using a variety of techniques and locations.

Study Design: The author admits in the report that the survey was "non-quantitative". In this sense, the design was not adequate to allow quantitative comparison of fish from exposed areas with those from reference areas.

References and Controls: The study did include upstream and downstream stations on the receiving streams and did compare the results to pre-operational data. However, because of the variety of fishing techniques used, a meaningful comparison between fish populations in exposed areas and reference areas cannot be conducted.

Study Methods: In order to make proper quantitative comparisons, the methods used must be consistent. As well, attempting to make any interpretation based solely on the numbers of fish present is probably inadequate. The description of the field and laboratory methods were very cursory.

Data Analyses: The numbers of each fish species captured at each sampling location were simply reported; no statistical analyses of the data was presented. No attempt was made to estimate catch-per-unit-effort or to make any interpretation about size or age distributions.

Overall, the fish population study was not adequate to make any conclusions beyond the mere presence of fish species in the receiving environment.

1.2.5 Fish - Tissue Residues

Another study (Report No. 346) examined the accumulation of fluoride in fish tissue and effects of fluoride on fish. Fluoride was an unregulated constituent of the mine effluent. Caged fish were exposed to the receiving waters and native fish were collected for tissue analyses.

Study Design: The design for the caged fish study was adequate in terms of sample size, sample stations and parameters measured. The study evaluated a considerable amount of water quality data but did not examine fluoride in sediments. This was probably a shortcoming.

Reference and Control Sites: Adequate for the caged fish study and the water quality data; some question about the mobility of native fish between reference areas and exposed areas.

Study Methods: Study methods were described in good detail.

Data Analyses: Data analyses was thorough and statistics (ANOVA) were used to determine differences between reference and exposure sites.

Conclusions: Although this study was limited in scope to the fluoride issue, it was adequate in identifying the accumulation of fluoride in fish tissue and assessed the effects of that accumulation on the fish.

1.2.6 Toxicity Testing

Standard rainbow trout acute lethality tests were conducted on samples of final effluent at least twice per year by the federal government and on an ad hoc basis by the company. The results of these tests are summarized in Table 1.4. The effluent was non-acutely lethal to the test fish in 18 of the 23 tests conducted (80%). Early in the mine's operation, an upset condition occurred and total suspended solids in the effluent exceeded regulated limits for a period of a few months. During this period, there were a few toxicity test failures (less than 50% survival in full strength effluent). Modifications to the waste water treatment system corrected this situation. In 1992, shortly after the mine ceased operation, there were a couple of toxicity test failures which were attributed to elevated concentrations of aluminum. More diligent control of pH levels through the waste water treatment system corrected this problem.

Study Design: Since the final effluent quality from the waste water treatment was quite consistent, the monitoring of that effluent for acute toxicity was intermittent in nature.

References and Controls: All of the laboratory toxicity tests were conducted following standard Environment Canada protocols with proper control tests.

Study Methods: All of the laboratory toxicity tests were conducted following standard Environment Canada protocols.

Data Analyses: The results of the toxicity tests were analysed as required by the standard test procedures.

Conclusions: The final effluent from the tin mine typically passed acute toxicity tests conducted on fish and daphnia. The toxicity testing only examined the final effluent and did not test receiving water samples. Only acute lethality testing was conducted; no sub-lethal or chronic tests were conducted. The monitoring was adequate to address the compliance requirements of the federal regulations and guidelines.

Table 1.4: Results of acute toxicity tests conducted on final effluent from the East Kemptville Tine Mine

Date	Test Species	Test Results
Dec. 10, 1985	Rainbow Trout	4-day LC50 = 83%
Feb. 28, 1986	Rainbow Trout	4-day LC50 = 22%
Feb. 28, 1986	*Daphnia magna*	2-day LC50 = 27%
May 28, 1986	Rainbow Trout	4-day LC50 > 100%
June 12, 1986	Rainbow Trout	4-day LC50 > 100%
Dec. 9, 1986	Rainbow Trout	4-day LC50 > 100%
Mar. 26, 1986	Rainbow Trout	4-day LC50 = 15%
June 30, 1987	Rainbow Trout	4-day LC50 > 100%
May 11, 1988	Rainbow Trout	4-day LC50 > 100%
Oct. 18, 1988	Rainbow Trout	4-day LC50 > 100%
June 12, 1989	Rainbow Trout	4-day LC50 > 100%
June 12, 1989	*Daphnia magna*	2-day LC50 > 100%
Oct. 24, 1989	Rainbow Trout	4-day LC50 > 100%
Oct. 24, 1989	*Daphnia magna*	2-day LC50 > 100%
May 14, 1990	Rainbow Trout	4-day LC50 > 100%
May 14, 1990	*Daphnia magna*	2-day LC50 = 49%
Oct. 5, 1990	Rainbow Trout	4 day LC50 > 100%
May 22, 1991	Rainbow Trout	4-day LC50 > 100%
Nov. 26, 1991	Rainbow Trout	4-day LC50 > 100%
May 4, 1992	Rainbow Trout	4-day LC50 = 62%
May 19, 1992	Rainbow Trout	4-day LC50 = 80%
Aug. 11, 1992	Rainbow Trout	4-day LC50 > 100%
Oct. 14, 1992	Rainbow Trout	4-day LC50 > 100%
Mar. 31, 1993	Rainbow Trout	4-day LC50 > 100%
May 13, 1993	Rainbow Trout	4-day LC50 > 100%
Apr. 14, 1994	Rainbow Trout	4-day LC50 > 100%
Nov. 30, 1994	Rainbow Trout	4-day LC50 > 100%

1.2.7 Zooplankton

A zooplankton survey was conducted in the tailings pond in 1986. Plankton net tows were conducted both during daytime hours and at night. Since the tailings pond does not represent the aquatic receiving environment for the mine, the results of the study are not of much actual use in determining impacts. The fact that the tailings pond did contain some limited numbers of common lentic zooplankton would indicate that the water quality in the pond could support some aquatic life. The study design was quite ad hoc and did not produce results that could be compared qualitatively to a reference area. Overall, this study was inadequate for evaluating any potential impacts to zooplankton.

1.3 Results of Aquatic Effects Assessment

1.3.1 Water Quality

Extent of Effects:
Temporal: Since the mine commenced operation (1986), an increase in hardness, pH and fluoride was noted in the receiving waters downstream of the final effluent discharge as compared to the reference stations at Pug Lake and upstream Big Meadow Brook.

Spatial: Changes in receiving water quality were observed at stations located 3.5 km and 5 km downstream of tailings discharge.

Magnitude of Effects:
At the assessment point 5 km downstream from the mine, a few parameters (fluoride, manganese, hardness) have exceeded the background (upstream) levels during the operational life of the mine; however, they have not exceeded the site specific water quality objectives for these receiving waters. The upstream reference stations provide excellent information for comparison with data from the monitoring locations downstream of the mine discharge.

8-year means: see Table 1.3

Background (upstream)

pH - 4.64	arsenic - 0.004 mg/L	copper - 0.01 mg/L
zinc - 0.02 mg/L	manganese - 0.02 mg/L	aluminum - 0.36 mg/L
hardness - 2.96 mg/L	fluorine - 0.11 mg/L	

Assessment Point (downstream)

pH - 4.86	arsenic - 0.004 mg/L	copper - 0.01 mg/L
zinc - 0.03 mg/L	manganese - 0.047 mg/L	aluminum - 0.37 mg/L
hardness - 15.17 mg/L	fluorine - 0.63 mg/L	

With the exception of the initial problem with suspended particulates and occasional elevated zinc levels, this mine operated in compliance with MMLER's and it's provincial industrial waste operating permit. The water quality objectives have been met for all parameters every year since 1987. Elevated levels of fluoride in the mine discharge necessitated the establishment of an objective for this parameter.

1.3.2 Sediment Quality

Methods/Tools:
Pre-operational baseline data for 9 stations were obtained as part of the EIA process. U.S.EPA Interim Guidelines for Dredged Sediments used for comparison. Sediment samples were analysed at EAG Laboratory. Triplicate samples were collected using an Ekman grab sampler.

Extent of Effects:
Stations 0.8 km and 5 km sampled in 1986.

Magnitude of Effects:
Arsenic - 23.3 mg/kg 0.8 km downstream of tailings discharge.
Arsenic - 6.9 mg/kg 5 km downstream of tailings discharge.

Remarks:
These values are for two downstream locations and are for samples collected before the mine operated but following construction activities at the site. These means exceed the U.S. EPA Interim Guideline. Duck Pond Brook sediments (0.8 km downstream) were adversely impacted by the discharge from the tailings pond and by siltation from construction activities on the site. The increase in arsenic levels in the stream sediments may be related to that siltation. Effects on Big Meadow Brook and Tusket River sediments were not detected.

1.3.3 Benthos

Methods or Tools:
Pre-operational baseline sampling at 7 stations Ekman dredge/Surber sampler.

Extent of Effects:
Sampling stations were located 0.8 km and 5 km downstream of the mine's discharge.

Magnitude of Effects:
Benthic analysis on two occasions in 1986 indicated that Duck Pond Brook 0.8 km downstream of tailings pond discharge contained a 50% decrease in benthos.

Five kilometres downstream sampling indicated a population density of twice that at control station.

Remarks:
It must be concluded that there has been a drastic reduction of invertebrates in Duck Pond Brook after it was altered by the construction of a tailings pond. This reduction is attributed to either siltation from the construction phase, the drying out of the brook for several months during construction or to some contaminants released during the operational phase of the mine or some combination of these factors. Further downstream sampling and analysis indicate no reduction in the number of taxa or organisms when compared with upstream control mechanisms.

1.3.4 Fish - Population Assessment

Methods or Tools:
Native fish were captured using a variety of methods: electro-fishing, G-Traps, Fyke nets and gill nets. Capture fish were identified as to species, sized and had their stomach contents examined.

Extent of Effects:
Not determined.

Magnitude of Effects:
Results were inconclusive as too few species were collected.

Remarks:
Four species were collected and some data was obtained from personal communications with mine employees. The stomachs of sampled fish were examined for the presence of benthos to obtain data on benthic presence at fish sampling locations.

1.3.5 Fish - Tissue Residues

Tissue residues and physiological indicators.

Methods or Tools:
- caged brook trout in receiving water downstream of mine
- native yellow perch and brook trout captured by electro-fishing gill net
- histopathological and X-ray analyses by Atlantic Veterinary College
- analyses of fluoride concentrations in skeletal tissue

Extent of Effects:
Skeletal tissue analysed for fluoride content from stations 3.5 km and 5 km downstream of tailings.

Magnitude of Effects:
Brook trout held in cages 3.5 km downstream of the mine had 600 μg/g wet weight of fluoride in their bones. (12 x background).

Brook trout held in cages 5.0 km downstream of the mine had 150 μg/g wet weight of fluoride in their bones (3 x background).

Remarks:
Hatchery raised speckled trout, when held in receiving waters downstream of the mine, demonstrated a significant increase (A=0.05) in the concentration of fluoride in their backbones over fluoride levels in reference fish. Resident yellow perch and specked trout from receiving waters had higher concentrations of fluoride in their backbones than fish from control stations.

Although fish exposed to receiving waters have been shown to accumulate fluoride in their backbones, no damage to their vertebral skeletons could be detected with pathological examination.

1.3.6 Toxicity Testing

There were 23 acute toxicity tests conducted on tailings pond effluent with rainbow trout and four tests conducted using *Daphnia magna*. The effluent was determined to be non-acutely lethal in the majority of samples. Acute toxicity was detected for brief periods in 1986 and 1992.

1.3.7 Zooplankton

A zooplankton net was used for both day and night sampling in the tailings pond in September 1986. The results were reported as similar to control station sampling carried out several kilometres upriver.

1.4 Discussion

Several environmental impacts were documented in the reports reviewed for the East Kemptville Tin Mine. In the receiving stream, Big Meadow Brook, increases in water hardness, alkalinity, pH, manganese and fluoride have been reported as a result of the discharge of waste water from the tailings pond. At the assessment point on the East Tusket River (5 km. downstream from the mine) pH, hardness and fluoride are elevated compared to the reference site. Site specific water quality objectives established for this location have always been met from 1987 to the present. These changes are can be related to the tin mine discharge.

Except for the fluoride, these increases do not appear to have had any adverse effect on the biota in these receiving waters and in fact, anecdotal information indicates that sport fishing for brook trout in Big Meadow Brook has improved since the mine commenced operations. This phenomenon is likely attributable to the increased pH and alkalinity in the brook due to the addition of lime to the effluent at the mine.

The benthic community appeared to be impacted in Duck Pond Brook but the study related these impacts to siltation from construction activities and to the closing off of the flow of water to this stream for several months when the tailings pond was being constructed.

Working Group 3 did not have a lot of information to draw on for this assessment. Only five reports were identified and screened into the AQUAMIN database. Two of these were pre-operational reports from the environmental assessment process and were not very applicable to the task. Most of these studies were one time surveys and would not be effective at identifying long term trends or impacts. The only benthic community surveys were conducted within the first year of the mine's operation. No follow up studies were performed.

The assessment conducted on resident fish populations in 1986 was of little value and again was only conducted the one time. Thorough surveys, conducted on a regular basis, would be more useful and could serve to quantify the anecdotal reports about increased brook trout populations downstream of the mine.

The study into the potential impacts of fluoride on fish in the mine's receiving waters spanned several years and clearly demonstrated that the fish were accumulating fluoride in their skeletal tissue but no adverse effects from that bioaccumulation were detected. No follow up studies have been conducted since 1991 to evaluate if the condition of fish in Big Meadow Brook has changed with respect to tissue fluoride levels or possible related effects. Even though the mine has been closed for 3 years, fluoride continues to be elevated in the effluent. In addition, no sampling has ever been conducted to evaluate whether concentrations of other contaminants such as lead, zinc or cadmium might be elevated in fish tissue from the area.

The water quality monitoring program conducted by the company under the guidance of the East Tusket River Monitoring Committee was very useful in this evaluation. The program has been in effect since before the mine commenced operation and through a careful selection of sampling locations and parameters, evaluations of changes in water quality can be readily made. In addition, because of the QA/QC program that has been followed for this program, the data base is quite reliable. Because this data is not published into a formal report, it was not screened into the AQUAMIN database. However, it was most useful for the purposes of this case study.

There was little information available on sediment quality in the receiving waters. Some data was collected during the pre-operational phase of the operation and one set of samples was collected in 1986. Although elevated levels of arsenic and chromium were reported by these two reports, they did not appear to be related to the mine effluent and no follow up studies were conducted.

The acute toxicity tests conducted on samples of final effluent have proven quite useful at detecting problems with effluent quality on two occasions: in 1986 when effluent quality was degraded and modifications had to be made to the tailings pond and water a treatment system; and in 1992, shortly after closure, when elevated levels of aluminum created some

concerns. As with the water quality data, the results of the fish toxicity tests are not reported in a formal manner and were not screened into the AQUAMIN database. The acute toxicity tests were only conducted twice per year and one could argue that more frequent testing would have been useful. The reviewers found no references to any sub-lethal or chronic toxicity tests conducted on effluent from this mine.

The use of a variety of sampling locations around this property makes the interpretation of the results more difficult. Sample locations must be defined and described thoroughly so that there are no doubts as to where samples were collected. Each of the surveys used unique sampling locations and there was very little consistency between studies. Any future studies should attempt to utilize the same sampling locations for benthic surveys, fish surveys, water quality sampling and sediment sampling. Simultaneous sampling in all of these media is also encouraged.

A network of groundwater monitoring wells was installed during the construction of the mine and these have been sampled on a regular basis since 1985. The East Tusket River Monitoring Committee assumed an advisory role for this program in 1993 and the monitoring data was compiled and reviewed in 1994 for the first time. No detectable impairment of groundwater quality was observed.

Overall, this mine operated for just over six years and had a good record of compliance with the *Metal Mining Liquid Effluent Regulations*. From the information available, the mine does not appear to have caused serious environmental impacts in the aquatic receiving environment as a result of its operations. Additional survey information to address the identified inadequacies of the monitoring results, however, would be required to confirm that conclusion.

1.5 Conclusions

1. With the exception of an initial problem with suspended particulates in 1986 and occasional elevated zinc levels, this mine had a good record of operating in compliance with MMLER'S and it's provincial industrial waste operating permit.

2. The final effluent from the mine was not acutely lethal to rainbow trout in 80% of the tests (18 of 23) conducted by Environment Canada between 1985 and 1994.

3. The receiving water quality objectives established for the East Tusket River were met and continue to be met for all parameters.

4. As a result of the fluoride in the mine's effluent, fish in the receiving waters accumulated fluoride in their skeletal tissue. No adverse effects on the fish from that bioaccumulation were detected.

5. The benthic community in Duck Pond Brook, the initial receiving stream, were impacted by construction activities when the mine was developed.

6. The benthic community surveys, the fish population surveys and the sediment sampling were not adequate to conclusively assess any potential long term effects due to the discharge of mine effluent.

7. Anecdotal information indicated that there was an improvement in brook trout populations in Big Meadow Brook due to increased alkalinity attributable to the mine effluent. Five kilometres downstream density of the benthos was twice the background; this may be attributed to the increased alkalinity from the mine effluent.

8. The setting of receiving water quality objectives and the establishment of the monitoring committee have been effective strategies for evaluating impacts from the mine's operations.

1.6 Recommendations

1. The water quality monitoring program (both surface waters and groundwater) should continue.

2. Additional studies into the current condition of the benthic invertebrate community and the fish community should be conducted.

3. Current contaminant residues (fluoride and metals) in fish residing in the receiving waters should be examined.

4. Sediment sampling should be collected in the receiving streams and analyses for metals and fluoride should be conducted

5. Due to identified concerns about the impacts from fluoride and aluminum, consideration should be given to establishing site specific effluent controls for these two parameters at this operation.

1.7 References

Environmental Applications Group Ltd. 1979. East Kemptville Project - Preliminary Environmental Programme. Prepared for Shell Canada Resources Ltd. AQUAMIN Report 713.

Environmental Applications Group Ltd. 1980. Environmental analysis 1980 - East Kemptville Project. Prepared for Shell Canada Resources Ltd. 33 p. AQUAMIN Report No. 621.

Niblett and Associates. 1986. Preliminary Evaluation: Benthos. Prepared for Rio Algom Ltd., East Kemptville Mine. 9p. + appendix. AQUAMIN Report No. 668.

Niblett & Associates Ltd. 1986. Post operational environmental monitoring program for East Kemptville Mine, Yarmouth, Nova Scotia. Prepared for Rio Algom Ltd. iv + 53p. + appendices. AQUAMIN Report No. 629.

Parker, W.R. 1992. Fate and effects of fluoride in aquatic ecosystems: a case study of a Nova Scotia tin mine. Submitted in partial fulfilment of the requirements for the degree of Masters of Environmental Studies at Dalhousie University. AQUAMIN Report No. 346.

Underwood, J.K., J.G. Ogden III, and D.H. Waller. 1989. Precipitation chemistry in Nova Scotia, 1978-1987. Water, Soil and Air Pollution 46:13-27.

Environment Canada. 1987. Recommendations for water quality objectives for the East Tusket River, Nova Scotia. Advisory Committee on Site Specific Requirements. Environment Canada, Atlantic Region.

2 Heath Steele Mines

2.1 General Mine Site Description

2.1.1 Geological Setting

The Heath Steele Mines (Heath and Stratmat) in northern New Brunswick (Figure 2.1) exploit complex massive sulphide ores which are found in volcanic and sedimentary rocks of the Tetagouche Group and are associated with porphyry and chloritic tuff (Report No. 294). The mines (Heath and Stratmat) are in the watershed of the Tomogonops River (Figure 2.2). Untreated site runoffs report to the upper North Branch Tomogonops (from Stratmat) and to the Little South Branch and South Branch Tomogonops (from Heath). Treated effluent from the tailings pond reports to the South Branch Tomogonops River. The Tomogonops tributaries combine to form the Tomogonops River about 15 km below Heath mine. The Tomogonops in turn joins the Northwest Miramichi 22 km downstream of the Heath mine and mill site.

Water in the Tomogonops tributaries above the mine site has a naturally elevated zinc content (up to 33 µg/L) and sediments may also carry elevated metals (Tables 2.1 and 2.2).

Above the mine site, stream water is characteristically soft (hardness levels of 7-34 mg/L) (Table 2.1). River pH (Table 2.1) varies seasonally, being in the range of pH 5-7, with pH tending to be higher in winter than in summer (M. Patterson, pers.comm.). Humic acid levels in the Tomogonops tributaries are twice those in the Northwest Miramichi (Report No. 734). The North Branch is typically much more brown in colour than the South Branch or Little South Branch Tomogonops (W. Fairchild and M. Patterson, pers. comm.).

There are no upstream industrial or municipal point-source discharges. In the watershed of the Tomogonops tributaries, however, there has been extensive logging and there is also one rock quarry south of the South Branch Tomogonops. These activities provide other sources of disturbance to stream habitats.

Northern New Brunswick experiences acid precipitation. Particularly at the time of spring runoff, this may add to the impact of acidic drainage from disturbed sulphide ores; any such cumulative impact, however, has never been documented.

Figure 2.1: Location of the Heath Steele Mine

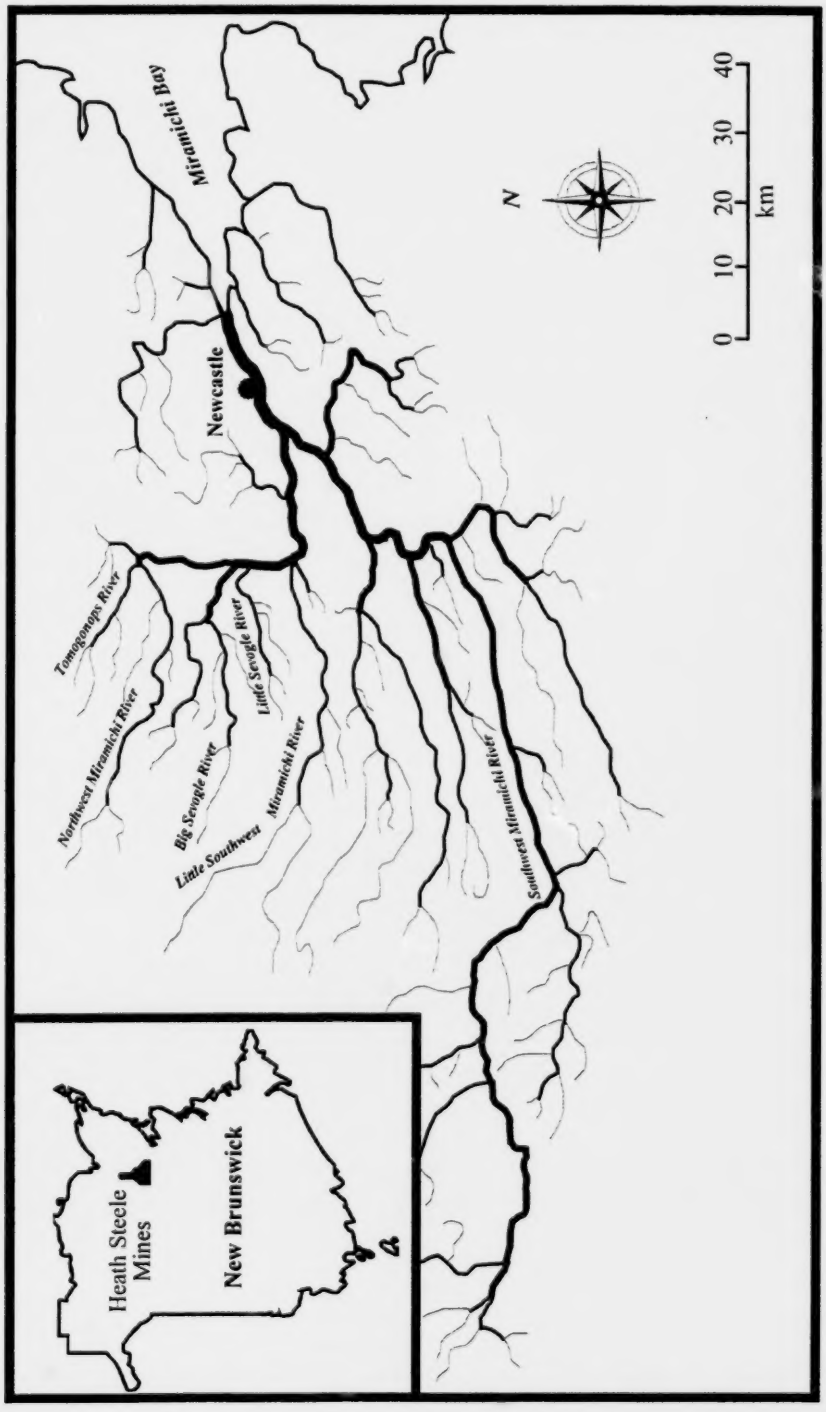


Figure 2.2a: The Tomogonops River system and the Northwest Miramichi River, showing sampling station locations

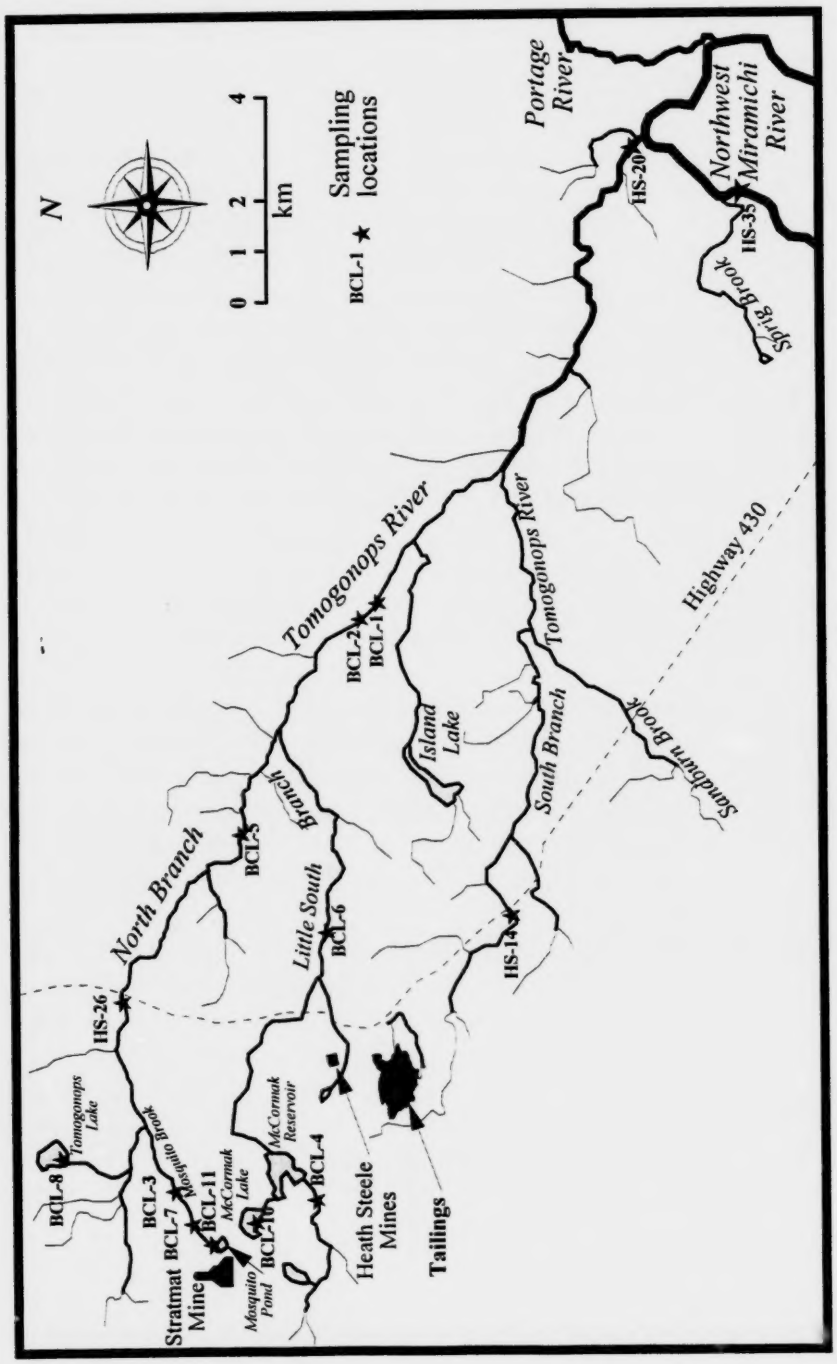


Figure 2.2b: The Tomogonops River system and the Northwest Miramichi River, showing sampling station locations

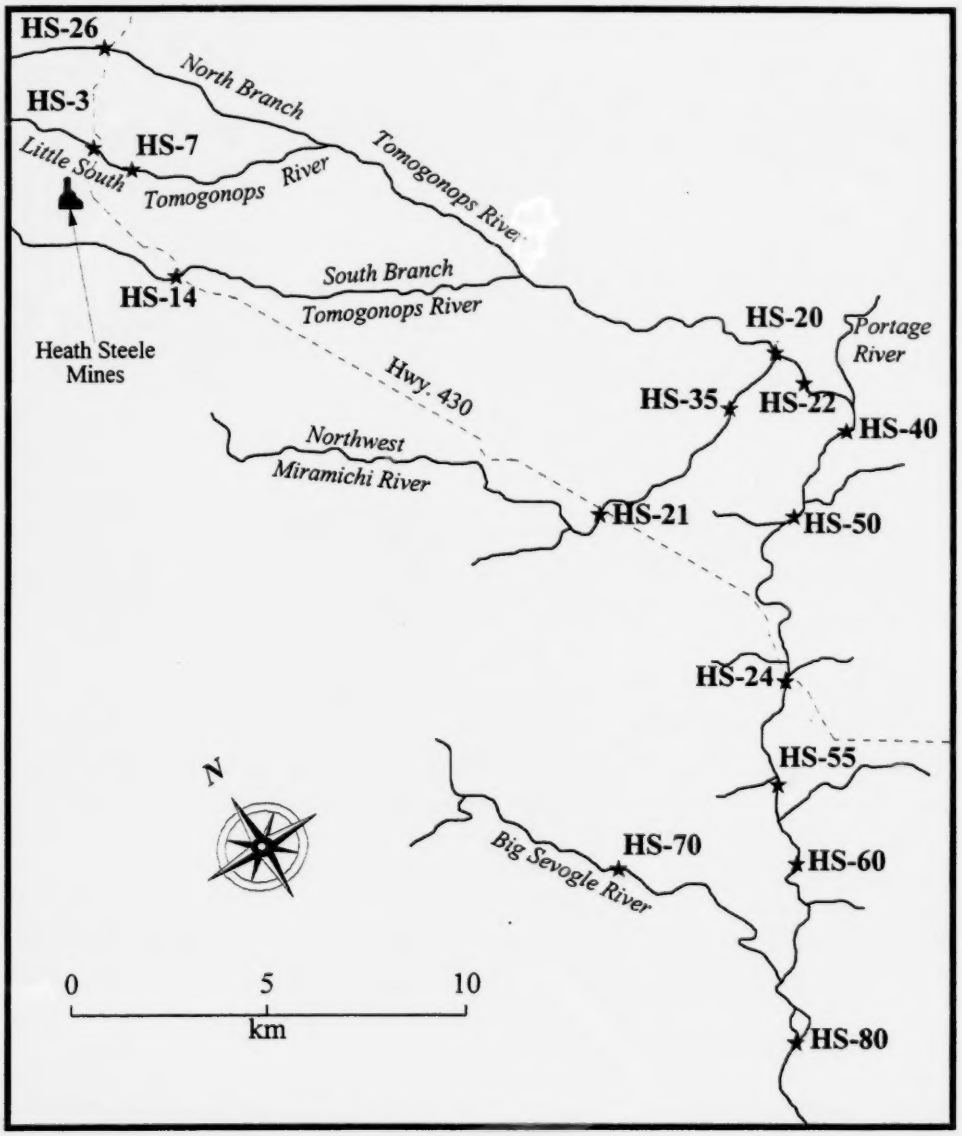


Table 2.1: Water chemistry at control sites (range, µg/L unless noted otherwise)

Site	Yr	Zn	Cu	Pb	Ni	As	Mn	Al	SO	Cd	SS	HD	pH
MCB	94	10	0.5	0.3	<1	0.5	ND	ND	ND	0.1	ND	10	7
BCL4													
SBT	81	20	0	<20	<5	ND	25	160	1	<5	ND	24	7
HS25		31					40	360	5			34	
NBT	72	7	<2	2	ND	ND	ND	ND	ND	ND	ND	6	6
H22	73	33	7	7								11	7
NBT	81	13	0	<20	<5	ND	20	164	ND	<5	ND	7	6
HS26		17					42					12	
MIN		7	0	0.3	<1	0.5	20	160	1	0.1	ND	6	6
MAX		33	7	<20	<5		42	360	5	<5		34	7
NWM	62	1	1	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
HS21	71	4	3										
NWM	72	3	<2	2	ND	ND	ND	ND	ND	ND	ND	10	7
HS21	73	19	5	6								18	8
NWM	84	10	<2	<2	ND	ND	10	600	3	<1	ND	ND	6
HS21			5	3			20	1100	4				7
NWM	90	20	10	10	ND	4	ND	ND	ND	ND	ND	ND	ND
HS21													
NWM	91	<10	<5	10	ND	ND	10	20	2	<2	ND	10	6
HS21		70	10	20			20		4			22	8
NWM	92	10	10	10	ND	2	ND	ND	ND	ND	0.4	14	7
HS21				30		5					36	64	8
NWM	94	4	0.3	0.8	<1	0.7	ND	ND	ND	0.1	ND	20	6
HS35													
MIN		1	0.3	0.8	<1	0.7	10	20	2	0.1	0.4	10	6
MAX		70	10	30		5	20	1100	4	<2	36	64	8

MCB = McCormack Brook above Little South Branch Tomogonops
SBT = South Branch Tomogonops
NBT = North Branch Tomogonops
NWM = Northwest Miramichi
SO = sulphate in mg/L
HD = hardness in mg/L
SS = suspended solids in mg/L.

References: Reports # 267, 268, 270, 271, 294, 393, 636, Busse 1974, Parker 1990, BEAK 1994

Table 2.2: Sediment quality at control sites (µg/g)

Site	Yr	Zn	Cu	Pb	Ni	Co	As	Cd	Mo	Cr
MCB BCL4	94	220	5	33	<5	5	1	2.7	ND	<5
SBT HS25	81	221	19	110	58	ND	<2	3.4	ND	ND
NBT HS26	84	76	21	23	ND	Nd	15	ND	ND	ND
NBT HS26	81	266	73	106	18	ND	<2	4.5	ND	ND
MIN		76	5	23	<5	5	1	2.7	ND	<5
MAX		266	73	110	58		15	4.5		
NWM HS21	91	63 149	9 19	24 29	20	19 1860	ND	<0.5 1	<1	24
NWM HS35	91	80	6	14	ND	1780	ND	<0.5	29	ND
MIN		63	6	14	20	19	ND	<0.5	<1	24
MAX		149	19	29		1860		.1	29	

References: Reports # 271, 294, 393, BEAK 1994

2.1.2 History and Type of Operations

Two mines and one concentrator are on the main Heath Steele site, for base metal mining and milling. Exploration in 1953 found two major ore zones. Mine development was initiated on this site in 1955, using a combination of open pit and underground shaft techniques. Milling began in 1956 (1500 tons/day). A second mine site, Stratmat, is located 4.5 km north-west of Heath beside Mosquito Pond. Stratmat was developed in 1987-1988, came into operation in 1989, operated until June 1993, and is now closed (Report No. 624). Mosquito Pond feeds Mosquito Brook, which reports to the North Branch Tomogonops (Figure 2.2).

Mill products are flotation concentrates containing zinc (54%), lead (50%), bulk (25-30% lead and zinc) and copper (22-24%). Mill capacity increased up to 3000 tonnes/d by 1972 and in 1994, operates at roughly 2,700 tonnes/day. Shut-down periods have been 1958-1960 (workings flooded; mill idle until 1962), 1983-1989, and June 1993 - October 1994.

The processing of ores involves crushing to six-inch rock and then milling to a fine powder (75% through #400 mesh). Water and reagents are then added to separate metals from the tailings. Process chemicals used (Nolan *et al.* 1992) are sodium carbonate (to raise pH to 9.6), sodium isopropyl xanthate, potassium amyl xanthate, methyl isobutyl carbinol, aryl phosphorodithionate, percol 351, copper sulphate, starch, Areo 3894, sodium tripolyphosphate, lime, sulphur dioxide (since 1981) and sodium cyanide (since 1983).

Originally, the mill site was serviced by rail. Today, all metal concentrates are trucked offsite: copper to the Murdochville smelter in Quebec, lead and bulk to the Belledune smelter in N.B., and zinc to Dalhousie Harbour, N.B. for shipping to Europe (Report No. 624).

2.1.3 Site Modifications and Efforts to Reduce Acid Mine Drainage

2.1.3.1 Heath Steele Mine/Mill Site

The Heath Steele mine site was historically host to more than eighteen separate piles of acid-generating waste rock, all but thirteen of which were milled and processed (Report No. 624). Under the terms of their provincial operating permit, contaminated water leaching from these piles must be collected and treated.

A legacy of past operations which poses an even more serious challenge to current environmental managers is the widespread use of acidic waste rock for construction of infrastructure in the 1950's (Report No. 723). This includes the rail bed, various road beds, and the mill site, all of which continue to contribute contaminants and acidity to the site runoff. The tailings dam, however, was not constructed of acidic rock.

Seepage and runoff of untreated AMD enters both the Little South Branch Tomogonops and the South Branch Tomogonops River.

Beginning in 1960 with the excavation of small settling ponds for lime-treated mine water, there has been a record of improved drainage and effluent treatment aimed at reducing environmental releases of AMD. Between 1964 and 1966 larger holding ponds were developed; these, however, could not contain heavy spring run-off (Report No. 734). Increasing heavy metal concentrations in the Tomogonops prompted stricter site-specific regulation by federal and provincial authorities in 1969. Subsequently, improved site drainage control reduced metals entering local rivers to less than 5% of previous levels (Report No. 393).

By 1981, buried hydrate wastes contributing to AMD had all been dug up and moved to the tailings pond (Report No. 649). Experimental covers for waste rock piles have been

undergoing testing since 1988. The covers have reduced infiltration by water, lowered temperatures in the piles and lowered oxygen concentrations; however, metal contaminated seepage continues (M. Patterson, pers. comm.).

In 1990 subsurface seepage was directed away from the clear water collection channel which drains to the Little South Branch Tomogonops. Improvements in EEM sampling and analysis procedures were also accomplished (Report No. 268). Prior to 1990, seepage from the north dam of the tailings pond entered the Little South Branch Tomogonops River via the clean water diversion channel. This is now collected and sent back to the tailings pond but the channel continues to carry some contaminants.

In 1991, a lined seepage collection pond was constructed for tailings dam west seepage, diverting it from the South Branch Tomogonops to allow it to be pumped back into the tailings pond.

In 1992, interception trenches and a collection pond were excavated to divert seepage away from the Little South Branch Tomogonops. The pH adjustment process was also refined so as to reduce soluble lead levels in the tailings pond effluent.

In 1993, the construction of an internal dam commenced in the Heath site tailings pond, to allow the flooding of the exposed tailings within the pond. The east side of the tailings dam was also partially reconstructed to reduce seepage.

In 1994, an acid-generating roadbed was removed from the north side of the Little South Branch Tomogonops, and an improved collection ditch was constructed.

2.1.3.2 Stratmat Mine Site

When the Stratmat mine went into operation in 1989, lime was mixed into the waste at a rate of 10 kg per tonne to inhibit or delay the onset of AMD. The treatment failed to suppress AMD problems, which were discovered in 1991, prompting construction of a seasonally-operated seepage collection pond.

AMD was discovered reporting to the North Branch Tomogonops via Mosquito Pond and Mosquito Brook in 1993. Further attempts to reduce metal loadings included routing the clean water diversion channel into the former mine pit. Water from this pit is pumped to the Heath site for treatment. In 1994, a seepage collection ditch was excavated to intercept contaminated shallow groundwater and pump this back to the pit. Zinc in the Upper North Branch Tomogonops in summer 1994 was at 30-40 µg/L, well below 170 µg/L, which is Beak Consultants' estimate of the minimum effect level (Beak 1994).

2.1.4 Wastewater Disposal

The control of untreated AMD on both the Heath and Stratmat sites is regulated through provincial permits which demand the collection and treatment of contaminated waters.

Mine water and surface water drainage are collected and pumped into the Heath minesite tailings pond, together with pH adjusted (limed) mill effluent. The alkaline conditions are required for precipitation of metals. The volume output from the pond varies seasonally, being maximal in spring (April to June) and fall (Oct). Average residence time was 3 to 4 days and average discharge was 3,780 gal (US) per minute in the early 1970's (Report No. 734). The average discharge in 1992 was 16.6 cubic meters per minute (Report No. 270). The effluent chemistry (Table 2.3) has varied over the years in response to changing ratios of mine water, site AMD, and mill wastes, as well as varying control over liming and pH levels.

Over 1992-1993, the tailings pond effluent was very hard (often over 800), alkaline (pH 9 - 12) and carried zinc (varying from 10 - 300 µg/L), lead (under 50 to over 1000 µg/L), and to a lesser extent, copper and arsenic (Table 2.3).

Table 2.3: Typical chemical content of treated effluent, as a range of monthly averages

Metal (MMLER limit)	Oct 1981	1982 min max	1987 min max	1988 min max	1989 min max	1990 min max	1991 min max	1992 min max	1993 min max	1994 min max
Zn µg/L (500)	154	30 297	37 3308	5 16603	9 14690	10 830	40 4240	10 300	10 310	30 280
Pb µg/L (200)	139	78 410	10 146	88 513	10 1100	140 3130	20 3800	30 1020	20 200	10 20
Cu µg/L (300)	16	8 19	10 136	14 584	5 629	10 70	10 70	10	10 20	10 30
As µg/L (500)	ND	ND	ND	ND	<1 21	1 5	1 20	2 5	3 4	3 4
SS mg/L (500	ND	ND	ND	ND	<0.5 46	3 19	7 39	1 11	2 6	1 7
pH (>6)	ND	8 11	8 11	5.5 10	7 12	9 12	9 12	10 12	9 11	7 12
# Months non-comp	0	1	4	8	8	10	8	6	1	0
Hardness mg/L	ND	ND	ND		962 1760	505 1133	607 911	560 832	ND	ND
Avg flow cubic m/min	14	ND	ND		10 60	3 39	1.5 41	10 22	ND	ND 22

References: Compliance reports, Reports # 267, 268, 270, 294.

Tailings dam effluent enters a ditch which reports to the South Branch Tomogonops River 1.5 km away. Some contaminated water seeps into the South Branch Tomogonops through the tailings dam.

There have been instances of excess suspended solids in the tailings pond overflow, carrying high lead and zinc, especially in high rainfall periods of spring and fall, or when pH was poorly controlled.

Documented spills of contaminated water occurred in 1960 (high zinc and copper from dewatering of mine) and in February 1991 (broken pipe released untreated mine water into the Little South Branch Tomogonops).

2.2 Quality and Quantity of Information for Mine Site

2.2.1 Description of Monitoring Programs

A program of water treatment and water quality data collection has been followed since the spill of mine water in 1960 (Report No. 734). Environmental monitoring of water chemistry has been performed since 1974 (Report No. 624). A benthic invertebrate monitoring program was initiated in 1981 (Report No. 294). Sediment sampling and electrofishing to assess fish populations are undertaken on an occasional basis.

Heath Steele Mines has an array of environmental monitoring stations for collection of water, sediment, fish, and/or benthic invertebrate samples (Figure 2.2). Control sampling sites are routinely included in each year's monitoring work. The value of the original control stations, however, has changed over the years as some have become impacted by AMD.

Control sites that remain valid are:

- BCL4 and BCL8 - (new sites in a stream and lake, respectively, which were selected for study in 1994) which provide background data for the Little South Branch Tomogonops and North Branch Tomogonops
- HS25 - on the Sandburn Brook, which joins the South Branch Tomogonops (not sampled in recent years)
- HS21 and HS35 - on the upper NW Miramichi
- HS70 - on the nearby Big Sevogle River

Former control site HS26 on the North Branch Tomogonops provided reliable background data until sometime between 1989 and 1991, when AMD from the Stratmat mine reached the North Branch. H12, HS1, HS2 and HS3, which were often cited as controls on the Little

South Branch Tomogonops in early studies, are all impacted to some degree by AMD from the Heath mine-mill complex, especially during heavy rains. Since 1987, sites on the Little South Branch Tomogonops may also have received treated effluent from the Stratmat mine, delivered via McCormack Lake (Figure 2.2).

The control sites in the upper Tomogonops tributaries have naturally elevated levels of zinc in the water, as well as elevated zinc and lead in sediments (Tables 2.1 and 2.2). The control site HS26 on the North Branch Tomogonops differs from impacted downstream sites in having macrophyte vegetation (Report No. 271), which complicates interpretation of changes in benthic fauna.

The control sites in the Northwest Miramichi are used to gauge the influence on this larger river of the combined effluents entering from the Tomogonops system. The NW Miramichi tends to have lower levels of zinc, copper and lead in sediments than found in the upper Tomogonops (Table 2.2).

Benthic fauna varies in diversity and density among the control sites (Table 2.4) and with season and year. Chironomids make up less than 45% (often <30%) of benthic organisms collected with Surber samplers.

Sampling sites impacted by AMD from Stratmat and/or Heath mines are (moving north to south):

- BCL11 - on Mosquito Pond
- BCL7 and BCL3 - on Mosquito Brook, between Stratmat and the North Branch Tomogonops River
- HS26 - upper North Branch Tomogonops River at Paynes Bridge
- BCL5 - on North Branch Tomogonops, above confluence with Little South Branch and, therefore, downstream of Stratmat drainage only (benthic plants present)
- HS1, HS2 and HS3 - on mine site, on Little South Branch Tomogonops (macrophytic plants present in stream at HS3 in 1991; Report No. 263)
- HS7 - downstream of mine, on Little South Branch Tomogonops
- BCL2 - on Tomogonops river between mouths of the Little South Branch and South Branch, above a railway culvert impassable to salmon
- BCL1 - on Tomogonops river between mouths of the Little South Branch and South Branch, below a railway culvert impassable to salmon

The tailings pond east overflow is sampled at station HS13.

Table 2.4: Benthic biota at control sites

Site	Year	Month	org/m²	no. genera	% chironomid
NBT	1981	Aug	1698	33	4
HS26		Oct	2743	33	2
SBT	1981	Aug	3774	39	11
HS25		Oct	5893	34	3
MCB	1994	Jun	21110	48	22
BCL4					
NWM	1981	Aug	494	25	13
HS21		Oct	826	43	2
NWM	1990	Nov	206	13	22
HS21					
NWM	1991	Jun	3416	49	27
HS21		Aug	1345	37	28
NWM	1992	Jun	1183	33	45
HS35		Oct	1076	33	15
NWM	1993	Jun	674	30	
HS35		Oct	1,000	32	38
NWM	1994	Jun	6073	32	6
HS35					

References: Reports # 262, 263, 272, 294, 711, BEAK 1994

Sampling sites downstream from both treated effluent and AMD drainage are:

- HS14 & HS18 - 2-4 km below tailings pond overflow on South Branch Tomogonops, by Highway 430 (These numbers refer to the same or closely related sites; the site numbers have changed over the years)
- HS8 - Tomogonops River below merger of North, Little South and South Branches
- HS20 - at mouth of Tomogonops (22 km)
- HS22 and 22E - on NW Miramichi below mouth Tomogonops (approx 24 km from mine)
- HS40 - approximately 26 km downstream on NW Miramichi
- HS50 - 30 km downstream
- HS24 - at Wayerton on NW Miramichi (36 km downstream)
- HS60 - approximately 40 km downstream
- HS80 - approximately 50 km downstream

The above sampling sites cover the full spectrum from the most impacted areas of the Tomogonops to the limits of the measurable metals contributed by the effluent and AMD.

In any given year, a subset of these sampling sites is monitored. The subset for monthly water chemistry data compiled by Heath Steele typically includes HS13 (effluent), HS1-3 (Little South Branch Tomogonops), HS18 (South Branch Tomogonops), HS20 (mouth of the Tomogonops), HS24 (NW Miramichi), and the control site HS21 (NW Miramichi).

Twice a year (typically late spring and early autumn, as determined by the province under their Certificate of Approval), Heath samples benthic communities, typically using sites HS3, HS14, HS20, HS22, and HS24, with control sites HS26 and HS21. Heath Steele is also responsible for quarterly testing of toxicity of their treated effluent and receiving environments using rainbow trout. Intermittently, more comprehensive sampling including benthic studies and fish population assessment as well as sediment and water chemistry is performed by consultants as part of a particular project. In-situ and laboratory toxicity testing on various species has also been performed occasionally by government scientists.

2.2.2 EEM Methods and Tools

2.2.2.1 Water Quality

Surface water sampling is by hand-held water bottle. New or (prior to 1991) Aqua regia-washed, 250 mL Nalgene bottles are used. Physico-chemical parameters (pH, dissolved oxygen, temperature, and conductivity), and stream depth and width are measured in situ, as well as a qualitative assessment of substrate type, stream flow, and vegetative cover.

Since 1990, total metal analysis is performed on 3 replicate samples for the purpose of QA/QC. For standard monitoring, only one sample per site is taken. Most analyses are for total metals. When dissolved metals are analyzed, the sample is filtered through a 0.45 micron Millipore filter. All samples are acidified with nitric acid prior to analysis by atomic absorption spectrometry or, in the case of lead, graphite furnace AA. Recent analysis for consultants' reports has been by inductively coupled plasma spectrometry. Hardness is calculated using atomic absorption analyses for calcium and magnesium.

QA/QC of Heath was tested in 1981 using spiked and unspiked water samples analyzed by three different laboratories (Heath, Brunswick and Noranda). Reproducibility was poor for copper at low concentrations at all 3 labs; poor for lead at Heath and Brunswick labs; good for zinc and manganese; variable for cadmium. Heath determined zinc and manganese accurately, but was less accurate for lead, copper and cadmium.

In 1988, analytical chemistry for standard EEM samples (i.e., water and sediment) was moved from the Heath Steele mine-mill complex to Brunswick Mining and Smelting in Belledune. The consequent improvement in reliability is shown by QA/QC results in 1990

(Report No. 267) and 1992 (Report No. 270), which show increased consistency among replicates (3 replicates per item).

A description of QA/QC measures followed by Beak Consultants in the course of special projects is provided (Report No. 271), although raw data from replicates are not published.

2.2.2.2 Sediment Quality

Fine sediments in the Tomogonops and Northwest Miramichi are restricted to small pockets and, therefore, must be collected opportunistically. Sediments are readily available from lakes in the watershed.

Stream sediments were collected by hand with a bottle or shovel; sediments in lakes and in the lower Miramichi estuary were collected using an Ekman grab sampler.

All samples were transferred to sterile Whirlpac bags and frozen for storage. Samples were subsequently dried at 60°C for 48 hours then sieved (using 841 µm in 1981, 177 µm in 1984) or left unseived (1991, 1994). After acid extraction, standard analytical methods (Water Quality Branch 1979) were performed. Loss on ignition tests were also performed to estimate organic content.

In 1981 (Report No. 294) replicates were analyzed at 3 labs to check reproducibility and accuracy. Heath's reproducibility was poor for copper, lead, arsenic, and cadmium, and good for zinc and nickel.

Accuracy at low levels was poor for copper, lead, and cadmium, and good for zinc and nickel. Heath's measurements of everything except copper were significantly different from (usually higher than) Noranda's.

2.2.2.3 Benthos

In most years, a Surber sampler was employed (0.09 m^2, 500 µm mesh collector, 2 samples per site) to collect benthic fauna from cobble and gravel bottoms. Sampling was performed by consultants in summer and early autumn. In 1994 the sampling regime was changed, and three samples were pooled to make up each of the two site samples.

On occasion (1972-73 and 1990), the colonization of crushed stone method (rock ball sampler) was employed.

For each sample, the number of organisms of each taxon was counted, and these data were used to determine various diversity indices. In Beak reports, taxonomic references were provided, together with QA/QC procedures (Report No. 271, Beak 1994). Taxonomic identifications have been typically made to the genus level except in the case of Nematoda, Acarina, and Turbellaria.

2.2.2.4 Toxicity

For regular tests of effluent toxicity, Heath Steele Mines, following Environment Canada protocol, exposes rainbow trout to pH-adjusted effluent (pH 7.7) in static conditions for 96 hours (Report Nos. 267, 271, 271). Toxicity of stream water to salmon was determined using 96 hour exposures in field cages (Parker 1990). Laboratory exposure tests were also performed to assess effects of stream water on fathead minnow and water fleas (Report No. 271). Toxicity of mine water was tested on salmon eggs (No. 271) and on rainbow trout at two different levels of hardness (Report No. 271).

2.2.2.5 Fish Populations

Fish populations were assessed by electrofishing (Report Nos. 294, 267, 265, BEAK 1994). In some cases all fish were caught by net, weighed and measured. In other cases, only a qualitative assessment (species presence/absence) was performed.

2.2.2.6 Fish Tissue Analysis

In one study, sampling of fish was by hand seine (Report No. 393). Metals analysis technique was not specified. Results were expressed on wet weight basis, without dry weight to wet weight ratio being determined.

In a second case (Ray 1978) salmon taken from the north-west Miramichi were measured and aged, then stored frozen until dissection. Tissue samples were analyzed for lead by AA spectrophotometry (graphite furnace) and results expressed on a dry weight basis.

In a follow-up to a spill of untreated mine water in 1991, Beak Consultants measured mercury, zinc, lead, and copper in fillets of four fish collected from the Northwest Miramichi River (Report No. 271). The methodology was not detailed.

2.2.2.7 Adequacy of Monitoring

The monitoring station network includes controls for both the Tomogonops and the Northwest Miramichi. Downstream stations cover a wide geographic area, including the portion of the Northwest Miramichi where metal concentrations return to background levels.

Baseline data for EEM were collected in 1981 (Report No. 294).

Control site data are most plentiful for HS21 in the Northwest Miramichi (Table 2.1). The Sandburn Brook (HS25) control site has not been sampled in recent years. The sampling distribution and controls should be reviewed for adequacy at regular intervals.

Not all stations were sampled in every year for all parameters; however, the data collected over time (especially data collected since 1990) was adequate for defining the magnitude and extent of effects on water chemistry.

The main problem with respect to water chemistry data appeared to be the poor reproducibility and accuracy of some early analytical data as evidenced by the QA/QC results of 1981. It is important to note that analysis for zinc was considered consistently reliable in both water and sediments. Trends in water quality through time of some less common metals are affected by early readings that tend to be overestimated.

Benthic sampling for numbers and diversity of fauna was restricted to 2 to 3 samples per site, which was considered by BEAK consultants to be inadequate for statistical purposes but good enough to indicate general levels of abundance and diversity, as well as presence or absence of metal-tolerant and intolerant species. The use of 500 μm collection nets restricted collections of benthic organisms to relatively large individuals. Of the two benthic sampling techniques, colonization was less reliable than Surber sampling, owing to high losses of equipment using the former method. The general trends in benthic abundance and diversity were similar regardless of sampling technique.

Benthic organisms on or in fine sediments were not documented. Monitoring of sediment chemistry was much less comprehensive than that for water chemistry and data were highly variable (Tables 2.5, 2.6 and 2.7). The sediment sampling performed to date has not added significantly to the understanding of potential impacts, because the degree of bioavailability of metals in these sediment habitats is unknown. Sediments accumulate in occasional patches; however, the actual percentage of bottom occupied has not been determined.

Sediment chemistry in potentially impacted lakes close to the Stratmat minesite deserves attention. Currently, research is being directed to defining bioavailability of metals in these habitats (M. Patterson, pers. comm.).

Table 2.5: Geographic extent of elevated metals in sediments of the Tomogonops and NW Miramichi Rivers over time. Furthest measured effect expressed as elevation over maxima of Tomogonops controls HS25 & HS26.

Metal	1981 (841 µm sieve)		1984 (177 µm sieve)		1991 (not sieved)	
Cu	22km	1x	36km	1.5x	24km	1.9x
Zn	24km	1.8x	36km	3x	24km	1.4x
Pb	2-4km	2.5x	22km	3x	24km	2.1x
Ni	2-4km	1.1x	no data		36km	1.9x
As	2-4km	1x	22km	1x	no data	
Mn	no data		no data		26km	1.7x
Cd	2-4km	1x	no data		2-4km	3x
Cr	no data		no data		36km	2x
Co	no data		no data		26km	2.9x
B	no data		no data		24km	1.7x

References: Reports # 194, 393, 271

Table 2.6: Sediment chemistry in South Branch Tomogonops over time, as measured 2 to 4 km downstream of tailings dam (HS14/HS18) compared to maxima recorded at controls on upper North and South Tomogonops (BCL4, HS25, and HS26, see Table 2.2) or (in the case of molybdenum) NW Miramichi site HS21.

Metal	1981 (841 µm sieve)	1984 (177 µm sieve)	1991 (not sieved)
Cu	3.5x	35x	60-88x
Zn	2x	6x	9-20x
Pb	2.4x	11x	12-20x
Ni	1x	no data	1x
As	1x	73x	no data
Mn	no data	no data	2-3x
Cd	1x	2x	1-2x
Cr	no data	no data	30x
Co	no data	no data	1-49x
Hg	1x	2x	2x
Mo	no data	no data	360x

References: Reports # 294, 393, 271

Table 2.7: Magnitude of effects of AMD on sediment chemistry in Little South Branch Tomogonops River. As measured at HS3/HS7 and compared with maxima from controls HS25/HS26/BCL1.

Metal	1981	1984	1991
Cu	7x	5.5x	9-30x
Zn	1.7x	1.2x	2.6-5x
Pb	2x	1.5x	2.2-5x
Ni	0.6x	no data	1x
As	1x	3.9x	no data
Mn	no data	no data	1x
Cd	1x	no data	1x
Co	no data	no data	1x
Cr	no data	no data	1x
Hg	1x	1x	1x

References: Reports # 294, 393, 271

Collection of fish population data was limited and sporadic (Report Nos. 294, 393, 267, 265, 642, 271). A 1994 study (Beak 1994) was more comprehensive than usual. The lack of electrofishing surveys in some stations impacted by the tailings pond effluent may be related to the difficulty of using some electrofishing equipment in very hard water. In addition, the return on investment in surveying waters that are realistically expected to be toxic to all fish, is poor. Now that water quality is improving, it makes more sense to monitor fish populations over a wider range of sampling stations.

Toxicity determinations were restricted to acute toxicity tests and there was no assessment of possible chronic or sub-lethal effects. Experimental animals used for toxicity tests were hatchery raised and, therefore, not acclimated to elevated metal concentrations. Changes in water chemistry with season, weather, etc. make it difficult to interpret short-term data. Occasional field tests cannot reveal the full range of potential effects (Clements and Kiffney 1994). In addition, laboratory and in-situ cage trials do not allow fish to find refugia that may be available to unconfined populations.

In 1994, the presence of fish in waters that had theoretically lethal metal concentrations underlined the need for cautious application of laboratory data to field conditions (Beak 1994).

Measurement of metals in fish tissues by Bailey (Report No. 393) was too limited to be

useful, and the lack of standardization for tissue water content invalidates the data. The study by Ray (1978) was methodologically sound, but limited to examination of lead levels in salmon. The Beak data (Report No. 271) are difficult to judge owing to a lack of methodological detail.

Benthic organisms and periphyton were not assessed for metal content. However, this type of analysis may be a better indicator of bioavailability than is ambient water chemistry (Kiffney and Clements 1993). Inclusion of such studies in the monitoring program becomes particularly important as improvements in water quality allows organisms at different levels of the food chain to encounter and accumulate non-lethal levels of metals. This situation could lead to food-chain transfers over a wider area than experienced in the past, possibly posing an increased threat of sub-lethal effects at higher levels of the food chain (St. Louis *et al.* 1993, Biguet *et al.* 1993).

2.3 Magnitude and Extent of Environmental Effects

Although pre-mine data is not available with respect to fish and benthic populations in the Tomogonops River system, it is widely accepted that, historically, this river hosted healthy populations of salmon and trout.

Today, potentially toxic metals enter this aquatic environment from the Heath Steele mine site in a variety of ways; the tailings dam overflow is not the only contaminated discharge. Because of the waste rock piles and infrastructure on the property that generate AMD, site runoff after rainfall, as well as seepage from swamps, springs, and through the tailings dam all deliver metals to both the Little South Branch Tomogonops and South Branch Tomogonops. The combined influence of AMD and treated effluent has had a dramatic negative effect on the Tomogonops River system (Report No. 734).

Untreated discharges are acidic (pH 3) and may contain high concentrations of zinc, copper, and lead. These are the only type of discharges affecting the Little South Branch Tomogonops River (Table 2.8).

Treated discharges (Table 2.3), which report to the South Branch Tomogonops (Figure 2.2, Table 2.9), are usually very alkaline, are extremely hard and carry metal contaminated suspended solids together with elevated dissolved zinc and lead. Copper concentrations are less than 10 µg/L. Toxicity trials using pH-adjusted effluent (pH 7.7) on trout (Report No. 267) indicate that the metals in the effluent are not acutely toxic. The effluent's extreme alkalinity has, however, rendered it lethal to trout in some years. In recent tests (1994), trout survived in effluent without pH adjustment because site operators worked to lower and stabilize the tailings pond pH (M. Patterson, pers. comm.).

Table 2.8: Magnitude of effects of untreated AMD on water quality in the Little South Branch Tomogonops River (HS3/HS7) over time (compared to controls HS25/HS26).

Metal	1972-73	1981	1989	1990	Feb 1991 (spill)	1991 (not Feb)	1992
Cu	10-175x	100-120x	14-185x	9-74x	1040x	1-3x	4-12x
Zn	35-550x	39-74x	44-219x	44-632x	6550x	3-6x	13-36x
Pb	2-6x	1x	1-7x	1-7x	10x	1x	1x
Ni	no data	1x	no data	no data	no data	no data	no data
As	no data	no data	1x	1-2.5x	66x	1-4x	1-4x
Mn	no data	76-87x	no data	no data	no data	8x	no data
Cd	no data	1x	no data	no data	no data	1x	no data
Al	no data	3-9x	no data	no data	no data	1x	no data
Sulphate	no data	7-18x	no data	no data	no data	1x	no data
Hardness	2-5x	1-2x	1-2x	1-1.6x	no data	1-7x	1-1.5x
pH	4-6	4-6	2-5	4-7	3	5-7	4-6

References: Reports # 636, 294, 266, 267, 262, 271, 268, 270, Busse 1974, Parker 1990.

Table 2.9: Magnitude of effects of treated effluent plus AMD on water chemistry of South Branch Tomogonops River over time. Max impact (HS14/HS18) compared to Tomogonops controls HS25 and/or HS26 (pre-1990).

Metal	1972-73	1981 and 1984	1989	1990 and 1991	1992
Cu	1-15x	15-44x	4-160x	1-5x	1-2x
Zn	2-25x	7-39x	2-199x	2-42x	2-9x
Pb	1-19x	4-44x	1-7x	4-84x	1-10x
Ni	no data	1x	no data	<2	no data
As	no data	no data	1x	1x	1x
Mn	no data	35-47x	35-155x	25-45x	no data
Cd	1-2x	<1-3x	no data	1x	no data
Al	no data	2x	no data	3-23x	no data
Sulphate	no data	no data	no data	156-348x	no data
Hard	40-55x	16-50x	2-38x	10-30x	10-21x
pH	6-10	4-7	5-11	4-11	5-10
SS	no data	no data	1-22x	1x	1x

References: Reports # 636, 294, 393, 266, 267, 262, 270, Busse 1974, Parker 1990

Process chemicals used in the Heath Steele mill are not believed to be present in the effluent at harmful concentrations (Nolan et al. 1992). Specific monitoring for cyanide has failed to detect significant levels of this chemical (M. Patterson pers. comm.)

The high pH of treated effluent has in the past caused precipitation of gypsum on benthic habitats in the Tomogonops and as far away as the confluence with the Northwest Miramichi (Report No. 271). Recent adjustments to the liming process have reduced this problem.

Lead precipitates from the water column and accumulates in the sediments close to the mine site. Zinc travels the farthest and is the main element of concern (Table 2.10).

The chemical influence of the Tomogonops can be tracked along the eastern shore of the Northwest Miramichi below the convergence of the two rivers. Elevated concentrations of metals are greatly reduced through dilution over the next 4 km. Elevated hardness and conductivity (Figure 2.2) are evident even at HS80, approximately 50 km below the Heath mine.

Water samples taken below the confluence of the Tomogonops with the Northwest Miramichi (36 km downstream of the mine) between 1962 and 1971 (Table 2.10) showed elevated copper, concentrations of which peaked in 1967 (7x), then declined to 1-3x background by 1972. Zinc levels also peaked in 1967 (80x), then declined. In the early 1980's, both copper and zinc were elevated at a distance of 36 km and increased to 9x and 31x background, respectively, at the mouth of the Tomogonops (22 km).

Since 1990, improvements in management of AMD and effluent have been complemented by improved reliability of analyses. Copper is now close to background at the mouth of the Tomogonops, except during exceptional circumstances such as the February 1991 spill. Major rain events following a dry period may also cause temporary elevations in metal concentrations (Report No. 270).

Zinc poses a chronic problem in the 1990's with concentrations being commonly 10-35x background at the mouth of the Tomogonops in 1992 - actual zinc concentrations of up to 350 µg/L (Report No. 270). After dilution in the Northwest Miramichi, zinc levels sometimes reach background at Wayerton (36 km), but may also be up to 9x background (Table 2.10).

Lead levels have historically not been a problem in far-field sites. Exceptions were during the 1991 spill, when lead exceeded background by 4x at 36 km distance, and during 1992, when over-liming of the tailings pond liberated unusual concentrations of lead and these showed up as far away as the mouth of the Tomogonops (1-3x background). This problem has since been reduced (M. Patterson, pers. comm.).

Table 2.10: Extent of effects of combination of treated effluent and AMD on water quality in Tomogonops and NW Miramichi Rivers. Furthest measured effect compared to (temporally matched when possible) Tomogonops control (#HS25, HS26 (pre-1990), BCL-4) or NW Miramichi control (HS21, HS35) as appropriate (see Table 2.1).

Metal	1962 - 1971	May Dec 1972	Aug&Oct 1981; Jun 1984	1990 1991	1991 Feb spill	1992*	Jun 1994
Cu	36km 2-7x peak 1967	36km 1-3x	36km 1-3x	22km 1x	36km 3-4x	22km 1-2x	22km 1-2x
Zn	36km 18-80x peak 1967	36km 3-11x	36km 1-6x	36km 1-3x	36km 100x 45km ≈45x	22km 2-10x 36km 1-3x	22km 4x
Pb	no data	24km 1-1.7x	22km 1-1.7x	24km 1x	36km 4x	22km 1-3x 36km 1-2x	22km 2x
Ni	no data	no data	24km 1x	22km 1x	no data	no data	no data
As	no data	no data	no data	24km 1x	36 km 2x	22km 1x	22km 2x
Cd	no data	no data	4 km 1-4x	24km 1x	no data	no data	22km 1x
Al	no data	no data	22km 3x	24km 3x	no data	no data	no data
Mn	no data	no data	36km 4-7x	22km 9-17x	no data	no data	no data
Sulphate	no data	no data	36km 4x	22km 37x	no data	no data	no data
SS	no data	no data	no data	22km 1x	no data	22km 1-2x	no data
Hardness	no data	no data	36km 1-4x	50km 2-3x	36 km 5x	no data	22km 12x

References: Reports # 636, 294, 393, 267, 268, 262, 271, 270, Busse 1974, Parker 1990, BEAK 1994
* Tailings pond over-limed (M. Patterson pers. comm.)

Elevations in copper, zinc, and lead levels in bottom sediments immediately downstream of the tailings pond (Table 2.6) can range up to 20 to 88x higher than maximum background levels. High concentrations of arsenic, cobalt, molybdenum, and chromium have also been recorded (Table 2.6). Levels of cobalt, molybdenum, and chromium may also be elevated in control sites on the upper Northwest Miramichi (Table 2.2). Downstream in the Tomogonops and Northwest Miramichi, sediments exhibit no more than 3x background levels in the metals measured (Table 2.5).

The presence of moderately elevated metal concentrations in Tomogonops river sediments is of unknown significance. The impact in the upper Tomogonops tributaries may be minor owing to the rare occurrence of soft sediments on the bottom. Recent work indicates that Stratmat area sediments are non-toxic and the bioavailability of metals is considered low (M. Patterson pers comm.). Adverse sub-lethal effects in trout have been attributed in part to exposure to metals-contaminated depositional areas in a Montana Superfund site (Pascoe *et al.* 1994). In that case, the sediments were toxic to a number of test species.

To date, no unusually high metal concentrations have been demonstrated in limited sampling of fish from the Northwest Miramichi (Report No. 393, Ray 1978, Report No. 271). There is evidence that levels of lead, cadmium, cobalt, manganese, selenium and thallium in young-of-the-year salmon sampled in 1993 did decline measurably moving from the mouth of the Tomogonops down the Northwest Miramichi (W. Fairchild, DFO, unpublished).

In situ toxicity testing (Parker 1990) and electrofishing surveys over the years show that within at least 2 to 4 km of the tailings dam runoff there are no fish, and in 1990 this was due to acute toxicity of the water of the South Branch Tomogonops River. The tailings effluent, when tested following Environment Canada's protocol, has been non-toxic consistently since the end of 1993. Judging from electrofishing surveys conducted prior to 1994, poor water quality down to the mouth of the Tomogonops deterred the entry of salmon into the river throughout the 1960's, 70's and 80's. At the river's mouth, 22 km downstream of the Heath mine-mill complex, dace was the only species found in moderate numbers prior to 1994. Occasional salmon were also caught (Report No. 294).

In 1981, a baseline survey disclosed brook trout, dace and chub present in the upper North Branch Tomogonops (above the influence of Heath minesite, but also, unfortunately, above a culvert that obstructs salmon migration). The control site on the upper Northwest Miramichi boasted salmon, as well as dace, chub, and stickleback. The upper South Branch (HS25) contained trout, dace, and chub. At HS3, within reach of limited AMD on the Little Branch Tomogonops River, brook trout, white sucker, dace, and chub all were found. In contrast, downstream of the AMD, below the confluence of the Little South Branch and North Branch Tomogonops, was devoid of fish (294) and a 1990 field toxicity test confirmed that the stream was acutely toxic (Parker 1990).

A spill of untreated mine water into the Little South Branch Tomogonops in 1991 was estimated to have killed up to 18% of salmon eggs at a distance 50 km downstream (Report No. 271). Subsequent studies showed benthic invertebrates recovered within months of the spill (Report No. 642). Reduced numbers of fry stage fish within 20 km of the spill were documented, but populations in the NW Miramichi appeared to be unaffected. Juvenile salmon (fry) were found in recovered densities in 1992 and 1993 and smolt from the 1991 year class were present in the expected numbers.

Laboratory tests in 1991 (Report No. 271), 5 to 7 days after the above spill, showed that water fleas could not survive in water taken from HS20 (mouth of Tomogonops River) and HS24 (36 km from mine, below the convergence with the Northwest Miramichi). Fathead minnows, in contrast, survived 96 hour exposures to water samples from these sites. Total zinc and copper at the mouth of the Tomogonops at this time were 930 µg/L and 30 µg/L respectively. At HS24, the concentrations were 300 µg/L (10x control site) and <10 µg/L (1x).

The density and diversity of benthic invertebrate populations in the South Branch Tomogonops has been reduced by 80-90% compared to control sites (Tables 2.4, 2.11) with Chironomids frequently making up the bulk (up to 100%) of a Surber sample (Table 2.11). Moving downstream to the mouth of the Tomogonops a trend of increasing diversity and density was evident (Table 2.12). At far-field sampling sites, chironomids occasionally occur in abundance (up to 52%). Similarly, they are sometimes elevated in numbers (up to 38%) at unimpacted sites. The proportion of Chironomids in Surber samples usually lies below 30% in control sites (Table 2.4), in the Little South Branch (Table 2.13) and below the mouth of the Tomogonops (Table 2.12).

In the Little South Branch, which receives only AMD, numbers of benthic invertebrates are often ten times greater than in the South Branch, but still less than half the number in control sites (Table 2.13). Similarly depressed (and highly variable) numbers are evident at least to the mouth of the Tomogonops (Table 2.12). Although the numbers of organisms may be depressed, the number of taxa in the Little South Branch Tomogonops is very similar to the number in the control site (Table 2.13).

In the South Branch Tomogonops the benthic community is relatively lacking in sensitive mayflies (Ephemeroptera), caddis flies (Trichoptera) and stoneflies (Plecoptera) (Table 2.11).

The biological influence of the mine was historically detected as far downstream as Wayerton, 36 km from the minesite, which for many years has had a slightly impoverished benthic community compared to upper Northwest Miramichi stations. Since 1992, no reduction in benthic biota has been noted in the mixing zone of the Northwest Miramichi.

In 1994, monitoring revealed that a dramatic improvement in fish populations had taken place. This could have been due to improved water quality stemming from mitigative work, although July-October 1994 was exceptionally dry, so that limited runoff could also have been a contributing factor (Environment Canada, McGraw Brook rainfall data). No impact on the diversity of benthic fauna was discernible at the mouth of the Tomogonops (Table 2.12) and salmon and other fish were present in moderate numbers (BEAK 1994). The Little South Branch Tomogonops was, as usual, devoid of fish. Although the stretch of stream below the confluence of the contaminated Little South Branch Tomogonops tributary remained relatively impoverished, salmon were found at BCL1 (above the confluence of the South Branch Tomogonops). This showed that salmon were no longer consistently deterred from entering the Tomogonops system. Five species, including brook trout, were found in the North Branch Tomogonops. The presence of culverts, rather than water quality, appeared to be the factor limiting access of salmon to the upper reaches of the Tomogonops above BCL1 (Beak 1994).

Table 2.11: Benthic fauna (i.e., relative diversity, relative density and proportion of chironomids) in the South Branch Tomogonops at HS14/HS18 (2-4 km below tailings dam). Numbers of taxa and organisms expressed in comparison with temporally-matched control (HS25).

	1981	1990	1991	1992	1993
no. taxa	0.1-0.2x	0.07x	0.2-0.3x	0.01-0.4x	0
no. organisms	0.01-0.07x	0.01x	0.01-0.02x	0.03-0.2x	0
% ch'mids	50%-77%	100%	50-91%	0-97%	

References: Reports # 294, 262, 263, 271, 270, 272, 711

Table 2.12: Extent of biological effects of combination of AMD and treated effluent, expressed in terms of relative diversity (# taxa i.e. genera), relative density (# organisms per unit area) and proportion of chironomids. Diversity and density at far-field sampling site (36 km in NW Miramichi or 22 km at mouth of Tomogonops) expressed as proportions of taxa or organisms found in temporally matched upper NW Miramichi or upper Tomogonops controls.

	1972-73**	1981	1984	1990**	1991	1992	1993	1994
no. taxa	36km 0.7-1x	36 km 0.4-0.8x	22km 0.4-0.5	36 km 0.6x	36km 0.9x	22km 0.2-0.3x	22km 0.2-0.4x	22km 1x
no. organisms	36km 1.8-1.9x	36 km 0.31-0.45x	22km 0.03-0.4x	36 km 0.5x	36km 0.8x	22km 0.1-0.3x	22km 0.05-0.2x	22km 0.2x
% Ch'mids		36km 0-12%	36km 2-6%		36km 52%	22 km 4%		22km 32%

References: Reports # 636, 294, 393, 262, 271, 263, 272, 270, 711, Busse 1974, BEAK 1994
** Artificial substrate samples (others are Surber samples)

Table 2.13 Benthic fauna (diversity, density and proportion of chironomids) of the Little South Branch Tomogonops, over time. Measured at AMD-affected sites HS3/HS7. Numbers of taxa and organisms expressed in comparison with control HS25.

	1981	1984	1990	1991
no. taxa	0.1 - 0.2x	1x	0.9x	0.9x
no. orgs	0.001 - 0.01x	1 - 1.5x	0.4x	0.4 - 0.6x
% ch'mids	6 - 50%	10 - 25%	30%	3 - 8%

References: Reports # 294, 393, 262, 263, 271

During the fish sampling program in 1994, it was noted that Northern Redbelly Dace were present at BCL3 where zinc concentrations attributed to AMD from Stratmat were theoretically lethal (1590 µg/L).

2.4 Costs and Benefits of Monitoring Program

Environmental monitoring costs money. It also saves money and helps to prevent and mitigate environmental damage.

The benefits of the monitoring program to Heath Steele have included:

- early warning of Stratmat seepage
- feedback for technical adjustments to tailings treatment
- early mitigation of unexpected and potentially expensive environmental problems
- possible avoidance of costs of prosecutions and litigation from other groups dependent on the river
- ability to document efforts in compliance with operating permit requirements
- better public relations, partnerships with anglers, etc.
- opportunity to attract government assistance with research in aid of continuous improvement.

Regardless of the benefits, it is reasonable to ask whether the monitoring program can be optimised for cost-efficiency as well as effectiveness. Monitoring programs need to be carefully targeted to maximize the useful information gained per unit of expenditure and to ensure that key environmental components are given proper attention.

2.5 Compliance Record in Relation to Effects

The mine must, under New Brunswick regulations, adhere to the MMLEG.

In 1982 the tailings pond effluent failed to comply with MMLER for lead in 1/12 months, whereas zinc and copper were within guideline limits. Other metals were not recorded.

In 1987, zinc exceeded the guidelines in 4 /12 of the months, while in 1988, there were exceedences in zinc, copper, lead and pH (8/12, 2/12, 2/12 and 1/12 respectively).

In 1989, lead was out of compliance in one month, zinc for 7 months. In all these cases, the major exceedence was for zinc in the month of April.

Over 1990 the tailings dam effluent was non-compliant for Pb in 10 months, and for zinc in

one. 1991 saw exceedences for lead (8/12), zinc (3/12) and suspended solids (2/12) with the greatest problems again occurring in the winter and spring i.e. January -April.

Recent improvements in tailings pond management and pH control have brought the effluent into general (but not yet absolute) compliance.

Over 1992, non-compliance for lead was recorded in 6 months, zinc in one month. In 1993, the effluent was out of compliance only once, for lead. Up to June of 1994, no exceedences were recorded for that year.

In the period that arsenic has been reported in the tailings dam outfall (since 1990) it has never exceeded MMLEG.

Problems with metals are often associated with higher than average TSS discharge. Treated effluent rarely drops below the regulated pH level of pH 6.

As of 1990, compliance data are also recorded for the Little South Tomogonops River, which receives acidic drainage from the mine site. Exceedences over MMLEG have been as follows:

1990: Zinc 12/12, copper 2/12, pH 10/12
1991: Zinc 12/12, copper 1/12, pH 9/12
1992: Zinc 11/12, pH 6/12
1993: Zinc 10/12, pH 6/12
1994: pH 2/6

Ongoing efforts to control acidic drainage to the Little South Tomogonops have not yet resulted in recolonization by fish (BEAK 1994).

The improvement in the compliance record for the tailings dam overflow has been accompanied an apparent improvement in the health of the receiving environment, as indicated by the recent discovery of salmon in the Tomogonops River system (BEAK, 1994). Additional monitoring is needed to determine the degree of improvement achieved through compliance for treated effluent and efforts to reduce acidic drainage.

As of 1994, Heath Steele Mines is an ARET site, and voluntary targets have been set for reduction of toxic discharges.

2.6 Discussion

2.6.1 Relationship Between Physical and Biological Effects

The environmental effects of Heath Steele Mines stem from the cumulative impact of treated effluent and untreated AMD.

The immediate receiving environment for treated effluent regulated under MMLER is the South Branch Tomogonops River, which does not support fish life over at least 4 km below the tailings pond. Benthic fauna are also depauperate along this stretch of the stream. The lack of fish could therefore be a function of direct toxicity, avoidance mechanisms, or lack of food.

The tailings pond effluent is alkaline, hard, and carries metal-bearing suspended solids as well as dissolved zinc and lead (Table 2.3). Water samples from June 1994 showed that in addition to elevated zinc, lead, and copper, the effluent also shows elevated nickel (7 µg/L), arsenic (2.2 µg/L), cadmium (0.4 µg/L), sodium (13 µg/L), potassium (1.7 mg/L), and sulphate (735 mg/L , 138x background).

On average over the year, concentrations of zinc in the stream in 1992 (Report No. 270) were almost double the concentration in the tailings pond effluent, lead levels were roughly one third of those in the effluent and arsenic was almost identical in the effluent and the stream. This reveals the augmentation of the effluent by AMD. The pH of the stream ranged from 4.8 to 10.1.

At sampling site HS 14/18 (Table 2.9), after dilution of effluent in the receiving stream and augmentation by untreated site runoff and seepage, average monthly zinc concentrations in 1992 (Report No. 270) were as high as 280 µg/L (9x background, April); lead levels reached 300 µg/L (10x, March); copper ranged up to 20 µg/L; arsenic remained below 5 µg/L; suspended solids peaked at 14 mg/L (March).

The volume flow of the South Branch Tomogonops is clearly insufficient to absorb and effectively dilute the combination of mine effluent and AMD, especially in spring runoff periods. At other times, release of effluent is not sufficient to counteract the pH effect of acidic drainage. In 1991to 1992, for instance, average monthly volume flow in the stream at HS14 ranged from 0.19 to 1.04 m^3/s (Report No. 270). Over this time, effluent discharge was 0.025 to 0.687 m^2/s with maximum discharge coinciding with spring high water levels in the brook. At some times, therefore, the volume flow of the stream can be made up of less than 10% effluent, while at other times it can be over 65%.

According to Sprague (1964) dissolved zinc can stimulate an avoidance response in salmon

at 53 µg/L or at as little as 6.1 µg/L in the company of 0.42 µg/L copper. Zinc at the mouth of the Tomogonops often exceeds this threshold.

The incipient lethal levels for salmon were 48 µg/L for copper and 600 µg/L for zinc (Sprague 1964b). Steelhead trout died in zinc of 66 to 670 µg/L, being less susceptible at lower pH, while copper was lethal at 2.8 to 66 µg/L (Cusimano *et al.* 1986). Rainbow trout avoided 5.6 µg/L zinc and died in 570 µg/L zinc (Sprague 1968). A lethal level for copper varied from 20 µg/L to 520 µg/L, with better survival in hard, alkaline water (Howarth and Sprague 1978). Eisler (1993) has demonstrated that increased water hardness reduces the effect of zinc toxicity.

In recent research by Beak (Report No. 271) using diluted mine water from Heath, acute toxicity for salmon and trout fry was found at 900 µg/L zinc for 96 hour tests and 450 µg/L in 14 day tests. Tests performed using Mosquito Brook and North Branch Tomogonops water in (Beak 1994) gave different results. In that case trout died at 110 to 320 µg/L zinc.

Copper in the range of 2.5 to 10 µg/L, which is typical of impacted reaches of the Tomogonops, is at the threshold of tolerance for many benthic insects (Heliovaara and Vaisanen 1993). Control sites may also have copper concentrations up to 10µg/L (Table 2.1).

The accepted safe level for lead in freshwater environments is 30 µg/L; sublethal effects on fish, however, have been documented in the range of 4 to 25 µg/L (Ray 1978).

In situ tests using caged salmon verified that the water at HS14/18 on the South Branch Tonogonops, as well as water below AMD seepage in the Little South Branch, was indeed lethal for salmon (Parker 1990).

There is no reason to suspect that lack of food is a factor in the lack of fish in the South Branch Tomogonops. Toxicity could be attributed to fluctuating pH, chronic zinc toxicity or acute lead toxicity, exacerbated by available copper. Zinc, which travels farther than lead and is present at higher concentrations than copper, appears to be the main factor influencing biota at stations below HS 14/18. The effects of zinc may be enhanced by copper concentrations exceeding 5 µg/L which occur to the mouth of the Tomogonops.

Chronic sublethal effects which may be linked to chronic exposure to low levels of metals (Heliovaara and Vaisanen 1993; Pascoe *et al.* 1994) have not been documented in the Tomogonops and cannot be ascertained from the current monitoring regime. Evidence from limited tissue sampling indicates no significant accumulation of metals in the flesh of salmon in the north-west Miramichi (Report Nos. 393, 271), although lead is found in relatively greater concentrations in liver and kidney tissues, and is present in higher concentrations in young salmon (parr) living in the Miramichi River than in larger fish (grilse) that have been

at sea for a year (Ray 1978).

A study conducted using freshwater shellfish and leeches in northern New Brunswick (Report No. bioaccumulate significant levels of copper and zinc, suggesting that food-chain transfers may be expected once herbivores return to formerly lethal reaches of contaminated streams.

Evidence in the literature (Kiffney and Clements 1993) indicates that a measurement of bioaccumulated metals in benthic invertebrates is a better indicator of bioavailability of metals than is ambient water chemistry.

Hardness in streams (at least up to 150 mg/L) is an ameliorating factor for metal toxicity (Report Nos. 735, 271), as are naturally-occurring humic acids (735). Neither of these has to date counteracted the lethal effects of mine effluent in the South Branch Tomogonops, but they may be protective further downstream, or as effluent quality improves.

The relative impoverishment of benthic fauna at sampling stations in the Tomogonops could, theoretically, be caused by exposure to metals dissolved in the water column or to metals in bottom or suspended sediment, provided these were bioavailable. The only evidence of bioavailability of metals in sediment comes from a study performed in the Miramichi estuary (Report No. 110). The sediments in question, however, are influenced not only by Heath mines but also by municipal sewage and pulp mills. The data on bioavailability are therefore not necessarily applicable to Heath Steele Mines' suspended solids.

The degree of variability in benthic fauna data underscores the need for conservative evaluation of such data, especially when samples are limited in number. Benthic fauna can change in response to many factors including vegetation, weather, streamflow, predation, competition, and season (Minshall *et al.* 1985). There is, therefore, little point trying to find trends over time by comparing the numbers of organisms present in different years or even in different months of the same year. Characterization of species present in terms of their chemical sensitivity gives more valuable information on environmental effects (Clements *et al.* 1992). Care must be taken, however, even in this type of interpretation, as "sensitive" groups, for instance the Trichoptera and Plecoptera, also contain metal tolerant species (Clements *et al.* 1992).

The consistently impoverished benthic fauna of the South Branch Tomogonops cannot be explained in terms of stream size, season, vegetation, etc. The predominance of metal-tolerant chironomids is consistent with the impact of toxic metals (Clements *et al.* 1992). As metal concentration in the water column diminishes downstream, benthic diversity improves. Marginal differences below and above the mouth of the Tomogonops in the Northwest Miramichi could be a factor of increasing stream size as well as of metal concentration. The

toxicity of river water to water fleas at zinc levels that were 10x background (Report No. 271) suggests that water chemistry can, especially in rainy months when suspended solids are elevated, have a negative effect on benthic diversity even 22 km downstream, where, in 1992, zinc could still be found at 10x background levels, or 350 µg/L (Report No. 270).

Fish population surveys provide presence-absence data for various species. Trends over time in terms of recolonisation of stream reaches are convincing indicators of biological response to changing water chemistry. This form of monitoring can also be used to check for the more obvious forms of chronic sub-lethal effects of some metals, for instance black tails and spinal curvatures indicating lead toxicity (Ray 1978). Electrofishing can also reveal unexpected populations of fish in waters that are theoretically lethal (BEAK 1994), providing information needed as a check on laboratory tests.

2.6.2 Do the MMLEG Adequately Protect the Environment?

The environmental effects of treated mine and mill effluents regulated by MMLEG include the elimination of fish populations and drastic reduction of benthic diversity in the South Branch Tomogonops with negative biological effects demonstrable to the mouth of the Tomogonops (22 km downstream), or further during episodes of increased runoff. The MMLEG do not yet adequately protect this receiving environment, in spite of the significant efforts in mitigation by Heath Steele.

Even though the effluent is currently in compliance with the MMLEG almost all the time, the receiving stream does not have sufficient volume flow to dilute toxic metals and consistently reduce associated high pH to a safe level. Therefore, although further improvements in biological populations may be possible downstream, attainment of background population densities and diversities may be impossible for the immediate receiving environment.

Revision of the MMLER and MMLEG to ensure that the volume flow and/or concentration of effluents is within the assimilative capacity of the receiving waters would help ensure that a situation such as we have at Heath Steele would not reoccur. It might also, of course, mean that some potential mine sites on acid-generating rock could not be brought into production cost-effectively, given current technological limitations and climatic realities.

Far-field effects, including the avoidance of the Tomogonops river system by salmon in most years, cannot be separated from the effects of untreated AMD reporting to the same system. Compliance with MMLEG cannot resolve this problem unless and until all site drainage is collected and treated. Currently, great efforts are being made to collect and treat all AMD by routing it through the tailings pond. Although serious gains have been made, the problem is

technically very challenging.

The MMLEG/MMLER could be strengthened to ensure that all possible measures be taken to collect and treat AMD. This amendment would make no difference to Heath Steele Mines, as it is already responsible for collecting AMD under its provincial operating permit. The benefit of harmonizing federal standards with this provincial requirement would be to provide national consistency - a reasonable minimum standard for all provinces and territories.

Elevated discharge of metals in spring and fall was associated with elevated suspended solids and difficulties in controlling pH during peak flow rate periods (Table 2.3, M. Patterson pers. comm.). Stricter regulation of suspended solids and pH or voluntary attention to treatment efficiency (as is currently practised by Heath Steele) could improve environmental protection.

Reduction of dissolved metals has natural physical limits with available technology; however, as shown in this case, the consistency of treatment can be improved, with positive results for the environment. The regulatory tool in this case could be higher penalties for non-compliance. The alternative is non-regulatory incentives for 100% compliance.

It has been suggested by some stakeholders that amendment of the MMLER/MMLEG could involve extending the regulations to cover a wider array of metals. In the case of Heath Steele, aluminum, manganese, cobalt, molybdenum, and chromium were the metals found in effluent which are not specifically regulated. However, negative impacts in the receiving waters have never been specifically attributed to these metals.

2.6.3 Can the EEM at Heath Steele Mines be Improved?

The standard annual monitoring program is adequate to characterize any changes in water quality stemming from mitigative actions or unexpected releases, provided that key sampling sites, including a valid control, are consistently included and that analytical results return promptly. The February 1991 spill, which went undetected for between nine and nineteen days, emphasises that a comprehensive monitoring scheme is only as good as the weakest link. The time elapsing between analytical results is effectively the duration of undetected spills one is willing to risk. In 1994, sampling was carried out every weekday. Heath Steele also has continuous conductivity measurements in the Little South Tomogonops as a form of early warning system (M. Patterson, pers. comm.).

Ecosystem health is reflected in general terms by the trends of benthic fauna diversity and density.

Targeted studies of invertebrate sensitivity and derivation of a suitable index of community sensitivity would improve the usefulness of the benthic monitoring (Clements 1992), and could lead to streamlining and reduction of long-term costs.

Fish population assessments performed on an occasional basis are a convincing reflection of changing ecosystem health, even though trout and salmon are not as sensitive to zinc, for example, as some invertebrate species (Moore and Ramamoorthy 1984, Trotter 1987). As water chemistry improves and fish reoccupy habitat, sampling of intermediate stations will be needed. It is likely that better use can be made of sampled fish (for instance the recording of physiological or gross morphological features) without a large increase in costs.

Targeted work on bioavailability of metals in benthic sediments would allow a rational decision to made regarding the usefulness of gathering further sediment chemistry data.

Now that animals are recolonizing reaches of streams that they previously avoided, attention should be given to bioaccumulation, food-chain movement of metals and possible sub-lethal effects.

2.7 Conclusions and Recommendations

The magnitude and extent of aquatic environmental effects of Heath Steele Mine are presented and referenced in Section 2.3 and Tables 2.6 to 2.13. These findings can be summarised as follows:

1. The metals of concern in the treated effluent mixing zone have historically included zinc, lead and copper. The pH was also a concern. Although not flagged as problems, levels of manganese, aluminum and sulphate also occur in concentrations that may be biologically significant (Table 2.9). Current concerns are zinc and lead, with the recent levels of manganese, aluminum and sulphate not known (Table 2.9). As of 1994, the effluent has passed acute toxicity trials even without pH adjustment. Previously, however, the alkalinity was lethal to fish.

   Recommendation: Working Group 8 should consider aluminum and manganese as potential additions to the list of parameters to be controlled in effluents on a mine site specific basis.

Rationale: Aluminum is known to cause chronic debilitation in fish owing to its effect on the gills and has also been linked to neurological problems in humans (Novaczek pers. comm.).

Manganese is persistent in aquatic systems. It has a relatively low toxicity but manganese chloride toxicity in mine waste polluted streams has been documented (Birge *et al.* 1980, Horne and Goldman 1994). Speciation of manganese in the case of Heath Steele is unknown.

Conventional treatment of mine waste water with lime to remove heavy metals may not be effective at removing aluminum and manganese. (R. Parker pers. comm.)

2. Elevated metal concentrations in non-MMLEG-compliant effluent are commonly associated with spring runoff and higher than usual suspended solids (compliance records) but releases of lead can also be associated with high pH (M. Patterson pers. comm.).

   Recommendations: Working Group 8 should consider whether the stricter regulation of suspended solids is a reasonable means of protecting the receiving environment from elevated metal concentrations.

   Incentives and regulatory tools for improved control over pH in tailings ponds should be considered, including the setting of a maximum allowable pH. Learning from the results at Heath Steele, setting the maximum allowable pH at 11.5, as currently in place in Ontario, would be beneficial. The regulations should deal with effluent limits; the methods employed to meet these limits should be the responsibility of the industry. In other words, the regulations should not specify particular control technologies or strategies to be used by the industry.

   Options that ensure adequate surface water management during spring or other high melt or precipitation periods should be sought, both for older facilities such as Heath Steele which were not designed for the collection of all surface drainage under high flow events, and for new operations. It should be kept in mind that the re-opening and operation of this older facility, in spite of its problems, has allowed for improvements to be made.

3. Prior to improvements in control of AMD and the general compliance of treated effluent, i.e. prior to 1992, and especially prior to the late 1980's, concentrations of copper, zinc, cadmium, aluminum, manganese and sulphate were elevated above 2x background (and in the case of zinc, manganese and sulphate, more than 10x) as far as 22-36 km downstream (Tables 2.1, 2.10). Except for during spill conditions, elevation

in lead concentration has been largely restricted to the immediate receiving environments (Table 2.8, 2.9).

4. Prior to 1992, salmon were excluded from all but the mouth of the Tomogonops River system and other fish populations also disappeared downstream of both the treated effluent and AMD discharges (Section 2.3). Benthic invertebrate populations were seriously depleted, with the effect apparent down to the river mouth (Tables 2.11 to 2.13). High concentrations of copper, zinc, lead, cobalt, molybdenum and chromium built up in sediments in the vicinity of tailings pond discharge (Table 2.6). In the Little South Tomogonops, affected by AMD alone, sediments accumulated lesser levels of copper, zinc and lead only (Table 2.7). At a distance of 24-36 km downstream, sediments bear 2-3x background levels of various heavy metals (Table 2.5). There is no information available regarding the bioavailability of these metals.

   Recommendation: Studies to determine the relationship between metal levels in sediments and the bioavailability of those metals should be put on the list of research needs.

   Rationale: This would allow sediment data collection to be rationalised.

   Recommendation: Working Group 8 should consider the addition of molybdenum and chromium as potential parameters for regulation on a case-by-case basis when elevated concentrations sediments are a concern.

   Rationale: Presence of these metals in sediments suggest their presence in the effluent. Both are potentially bioaccumulative (Eisler 1989, Calamari *et al.* 1982). Further research into the bioavailability and the toxicity of these metals is warranted.

5. The most serious deviations from background water quality have been apparent in zinc concentrations, which, while showing improvement, still can reach 4x maximum background levels at a distance of 22 km (Table 2.10). Zinc is therefore the current element of primary concern for far-reaching effects (Table 2.10).

6. Elevations in copper have historically been apparent as far as 36 km downstream. Copper levels are now within 1-2x background, both at 22 km (Table 2.10) and below the treated effluent outfall (Table 2.9). However, copper continues to be elevated in the AMD-affected Little South Tomogonops (Table 2.8).

7. Water chemistry concerns, with respect to AMD discharge into the Little South Tomogonops, have included elevated levels copper, zinc, arsenic and manganese and depressed pH. Although under better control since 1992, these remain well above

background levels (Table 2.8).

8. Effects on water chemistry and biota result from the combination of treated effluent discharge and acid mine drainage (Parker 1990). The problem is due to the receiving stream's lack of sufficient assimilative capacity for dilution of metals and reduction of alkalinity.

   Recommendation: Working Group 8 should look at the power of MMLER to ensure the collection and treatment of AMD and determine whether/how the regulation can be strengthened in this regard.

   Rationale: Although the MMLER theoretically recognizes AMD as a regulated discharge, it is the provincial regulations that bring this into effect in practical terms. This leaves open the possibility that in some provinces AMD could be given less than adequate attention.

   Recommendation: Working Group 8 should deal with the problem of ensuring that effluents are within the carrying capacity of the receiving waters, and consider whether this can be addressed at a federal level. In this case the option of regulating by means of water quality targets in or below the mixing zone deserves consideration.

9. EEM of tributaries affected by AMD from the Stratmat site reveal apparently healthy dace populations in a stream where zinc concentrations were theoretically lethal to rainbow trout.

10. In toxicity tests, it was determined that water fleas were much more sensitive than fathead minnows as indicators of downstream water quality (Report 271).

    Recommendation: In developing a framework for EEM or in recommendations regarding toxicity tests, there should be recognition of the potentially wide variance in sensitivity of locally-available indicator species.

11. The EEM program in place is adequate to characterize changes in water quality stemming from mitigative efforts or unexpected releases, provided that valid controls are consistently sampled and analytical results are promptly returned.

    Recommendation: In designing a national EEM program, Working Group 7 should ensure that feedback of data are timely and that proper control sites are included in every cycle of EEM.

12. The EEM program would benefit from additional sample sites for tracking

improvements in the extent of fish populations.

Recommendation: Working Group 7 should develop a framework for EEM that is flexible and responds to changing information needs so as to provide operators with optimal guidance and reward regarding improved performance.

13. The EEM program includes benthic sampling which, in the absence of sufficient replication, cannot provide definitive statistical proof of effects; however, it does provide useful data for a weight-of-evidence approach to determination of effects.
Recommendation: A national EEM program, if it is to include monitoring of benthic invertebrates, should recognize the limitations of the data and seek to balance costs with the usefulness of the data.

    Working Group 7 should consider alternative approaches to dealing with benthic invertebrate data, such as development of community sensitivity indices (Clements 1992), which could reduce costs and maximize the usefulness of benthic community data.

14. The EEM program includes sediment sampling which is difficult to interpret owing to lack of information about bioavailability of metals in sediments and the extent of depositional areas in various habitats.

    Recommendation: A national EEM program, if it is to include analysis of sediments, should also call for the measurement of bioavailability of such metals in the first EEM cycle, so that any further efforts can be rationalised.

15. The EEM program does not effectively monitor sublethal, physiological effects in fish or the effects of long-term loading of metals into estuarine systems downstream.

    Recommendation: In developing a framework for EEM and identifying research needs, Working Group 7 should provide guidance on the approach to cumulative and chronic effects monitoring.

    Research on bioaccumulation by periphyton, food-chain movement of metals and sub-lethal effects would provide useful information for the interpretation of EEM data. When fish population data are collected, consideration should be given to the collection of physiological health data at the same time, to maximise information gain per unit effort.

16. The EEM efforts at Heath Steele have detected recent improvements in the receiving environment. Salmon can now be found in streams above the effluent discharge,

meaning that they are no longer consistently deterred from entering the river system. However, fish and benthic populations continue to be seriously depleted in the tributaries that directly receive treated effluent and AMD, in spite of the fact that the treated effluent is not, by itself, acutely toxic.

Recommendation: In developing a framework for regulation and monitoring, working groups should not place undue importance on the predictive value of acute toxicity testing.

Rationale: In some cases, as seen in this case study, theoretically lethal metal concentrations can be compatible with presence of fish in the field. On the other hand, effluent that is non-lethal in laboratory tests may, either due to sub-lethal effects, effects on other species or augmentation by site AMD, continue to damage the receiving environment.

2.8 References

References from AQUAMIN database, by report #, in order of year

735 Base metal mine water pollution on the Nepisiguit River, New Brunswick. (R.H. Cook and R.A.W. Hoos, Dept. Fisheries & Forestry Devel. Br., 1971)

636 1972 New Brunswick Mine Water Quality Monitoring Program Data (R.A.W. Hoos, J.R. Machell & P. Hawkins, 1973)

?1973 New Brunswick Mine Water Quality Monitoring Program Data. (Surveillance Section, EPS, 1974).

662 Base metal mine waste management in northeastern New Brunswick (Montreal Engineering Co Ltd 1973)

543 The toxicity of raw and treated drainage from Heath Steele Mines, N.B. during period of September - October 1974. (P.G. Wells, E. Pessah and W.R. Parker, ESP 1974)

734 Acid drainage control and water treatment at Heath Steele. (Busse, E., 1974. Bulletin, Vol. 67 No 751, pp. 54-61)

620 Thiosalts and their measurement in mine mill effluents of northeastern New Brunswick (Pollock, T.L., H.S. Samant and D.A. Lord, Fisheries and Environment Canada 1977)

723 Report to Heath Steele Mines Ltd concerning investigations into the sources of contamination entering the Little South Tomogonops River (Montreal Engineering Co. Ltd., 1979)

294 Baseline Ecological Survey for Heath Steele Mines Limited, August-October 1981 (C.S. Wood)

649 Atlantic region base metal mine status report 1981. (G.F. Westlake, Water Pollution Control Div. 1982)

110 Trace Metal Association with Suspended Particulate Matter Due to Dredging of Miramichi Estuary, NB, Canada (S. Ray & S.D. McKnight, 1983)

111 Bioavailability of Cu, Zn, Cd and Pb from Contaminated Sediments by Bottom Dwelling Marine Organisms (S. Ray & M.R. Peterson, 1983)

393 A Heavy Metal Study of Three River Basins in Atlantic Canada Influenced By Mining Activities (H.S. Bailey, 1988)

624 Heath Steele - Stratmat Closure Plan (Heath Steele, 1988)

637 The use of a biomonitoring system in acquiring baseline water quality data for a metal mining operation in northeastern New Brunswick. (T.A. Smith for CEARC, 1989)

266 Surface Water Monitoring - 1989 Annual Report (I. Spice, 1990)

267 1990 Annual Report : Environmental (J. Duncan, 1990)

262 1990 Benthic Studies (Harris Industrial Testing, 1990)

269 Stratmat and N-5 Properties Baseline Soil Sampling Program (J.Y. Gagnon, 1990)

112 Relationships Between the Partitioning of Lead in Sediments and its Accumulation in the Marine Mussel, Mytilus edulis Near a Lead Smelter (B.P. Bourgoin et al., 1991)

268 Heath Steele Mines Environmental Report 1991 (Heath Steele, 1991)

263 Benthic survey of the Tomogonops and Northwest Miramichi Rivers - 1991 (Heath Steele Mines 1992)

271 Environmental surveys of the Northwest Miramichi River following a discharge by Heath Steele mines - 1991 (BEAK Consultants Ltd 1992)

697 The Miramichi River Industrial Point Source Report October, 1991 (Industrial Programs Section, N.B. Dept. of Environment)

265 Survey of Juvenile Atlantic Salmon in the Northwest Miramichi River, 1992 (BEAK Consultants)

270 Heath Steele Mines Environmental Report 1992 (Heath Steele)

272 Benthic Survey of the Tomogonops River System and the Northwest Miramichi River 1992 (Hare Fisheries & Env. Consult.)

711 Heath Steele Mines Environmental Review October 19, 1993 (Heath Steele, 1993)

653 Benthic Survey of the Tomogonops River System and the Northwest Miramichi River - 1993 (Hare Fisheries & Environmental Consultants, 1994)

642 Environmental studies of the north-west Miramichi River, 1991-93 following a mine water discharge (BEAK Consultants Ltd 1994)

References from outside of AQUAMIN Database, listed alphabetically by author

Beak Consultants Ltd 1994. Environmental survey of the Tomogonops River, 1994.

Birge, W.J., J.A. Black, A.G. Westerman and J.E. Hudson. 1980. Aquatic toxicity tests on inorganic elements occurring in oil shale. In: C. Gale (Ed.) Oil Shale Symposium: Sampling, Analysis and Quality Assurance, March 1979; EPA-600/9-80-022, U.S. EPA, Cincinnati, OH:519-534 (U.S. NTIS PB80-221435)

Calamari, D. *et al.* 1982. Toxicokinetics of low levels of Cd, Cr, Ni, and their mixture in long term treatment on *Salmo gairdneri* Rich. Chemosphere. 11:59-70.

Clements, W.H., D.S. Cherry and J.H. Van Hassel. 1992. Assessment of the impact of heavy metals on benthic communities at the Clinch River (Virginia): Evaluation of an index of community sensitivity. Can. J. Fish. Aquat. Sci. 49:1686-1694.

Clements, W.H. and P.M. Kiffney. 1994. Integrated laboratory and field approach for assessing impacts of heavy metals at the Arkansas River, Colorado. Environmental Toxicology and Chemistry 13(3): 397-404.

Cusimano, R.F., D.F. Brakke and G.A. Chapman. 1986. Effects of pH on the toxicities of cadmium, copper and zinc to steelhead trout. Can. J. Fish. Aquat. Sci. 43:1497-1503.

Eisler, R., 1989. Molybdenum hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildl. Serv. Biol. Rep. 85 (1.19). 61 pp.

Eisler, R. 1993. Zinc hazards to fish, wildlife and invertebrates: a synoptic review. Contaminant Hazard Reviews, Rep. No. 26. U.S. Dept. of Interior, Fish and Wildlife Service, Biological Report 10. Washington DC.

Heliovaara, K. and R. Vaisanen. 1993. Insects and pollution. CRC Press Inc., Boca Raton, Fla., pp. 183-200.

Horne, A.J. and C.R. Goldman. 1994. Limnology, second edition. McGraw-Hill, Inc., 576 pp.

Howarth, R.S. and J.B. Sprague. 1978. Copper lethality to rainbow trout in waters of various hardness and pH. Water Research 12:455-462.

Kiffney, P.M. and W.H. Clements. 1993. Bioaccumulation of heavy metals by benthic invertebrates at the Arkansas River, Colorado. Environmental Toxicology and Chemistry 12: 1507-1517.

Minshall, G.W., R.C. Peterson Jr. and C.F. Nimz. 1985. Species richness in streams of different size from the same drainage basin.

Moore, J.W. and S. Ramamoorthy. 1984. Heavy Metals in Natural Waters: Applied monitoring and Impact Assessment. Springer-Verlag Publishers, New York Inc.

Nolan, Davis & Assoc. 1992. Inventory of mining chemicals in Atlantic Canada, assessment of their fate and environmental impact. Environment Canada file # H91499.

Parker, W.R., 1991. An In-situ Study of the Acute Toxicity of the Tomogonops River System to Yearling Atlantic Salmon. Environment Canada, Environ. Control Br., Conservation & Protection: 13pp.

Pascoe, G.A., R.J. Blanchet, G. Linder, D. Palawski, W.G. Brumbaugh, T.J. Canfield, N.E. Kemble, C.G. Ingersoll, A. Farag and J.A. DalSoglio. 1994. Characterization of ecological risks at the Milltown Reservoir - Clark Ford River sediments superfund site, Montana. Environmental Toxicology and Chemistry 13(12): 2043-2058.

Ray, S., 1978. Bioaccumulation of lead in Atlantic salmon (Salmo salar) Bull. Environ. Contam. Toxicol. 19:631-636.

Sprague, J.B., 1964a. Avoidance of copper-zinc solutions by young salmon in the laboratory. J. Water Pollut. Control Federation 36(8):999-1004.

Sprague, J.B., 1964b. Lethal concentrations of copper and zinc for young Atlantic salmon. J. Fish. Res. Bd Canada 21(10):17-26.

Sprague, J.B., 1968. Avoidance reactions of rainbow trout to zinc sulphate solutions. Water Research, Pergamon Press, Vol.2:367-372.

Trotter, D.M. 1987. A Review of the Aquatic Effects of Metal. Prepared for the Water Resources Branch, Ontario Ministry of Environment

3 Hope Brook Gold Mine

3.1 Introduction

3.1.1 Site Description

The Hope Brook gold mine is located in south-western Newfoundland (Figures 3.1 and 3.2). Hope Brook Gold Inc., a subsidiary of BP Resources Canada, began commercial mining in May 1987. The ore body is hosted by felsic rocks. It occurs within a tripartite alteration zone comprised of an alumina-rich zone with 2-5% pyrite, 200 m in width; a quartz-rich ore-zone 60-80 m wide containing 5% pyrite, between 0.1 and 2% copper (as chalcopyrite) and elevated levels of antimony and silver; and a silicified pyritic zone, 75 m in width containing between 5 and 25% pyrite. The gold occurs as very fine free grains, less than 200 μm in size. Copper grades increase with depth. The ore reserves were estimated prior to the start of mining at 11,200,000 tonnes containing 4.5 g/t gold.

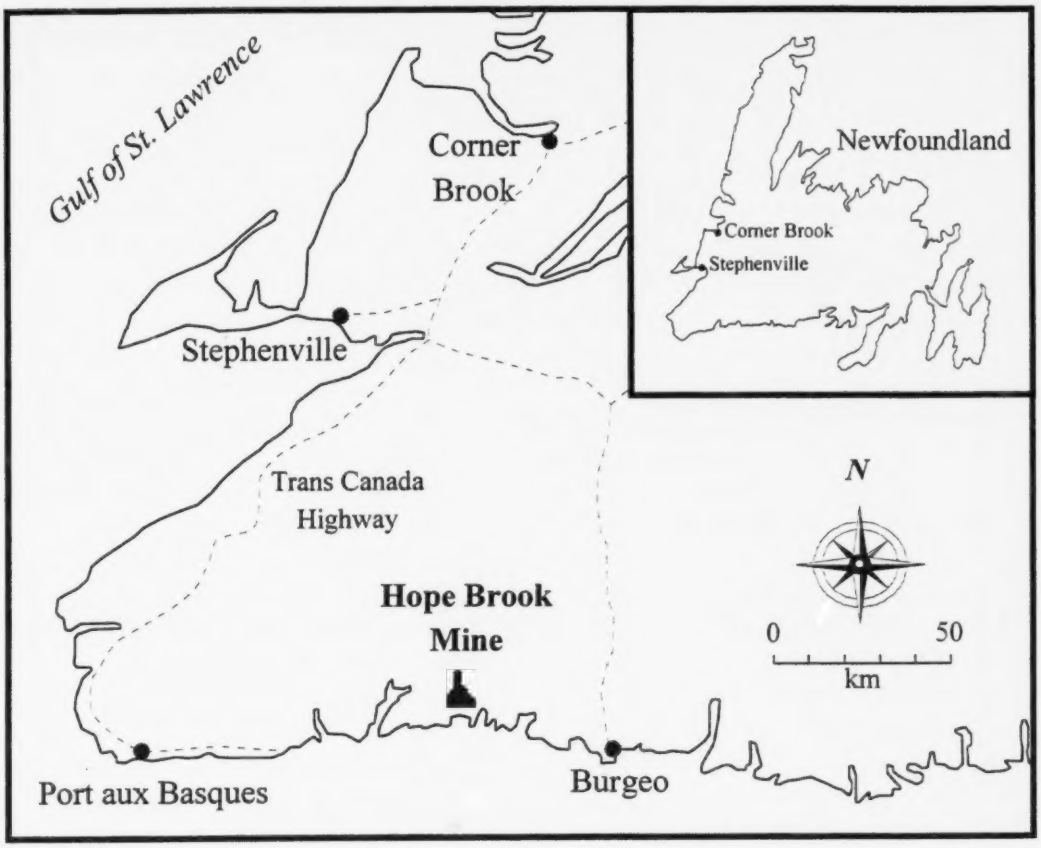
3.1.2 History and Type of Operations

Initial production of about 800,000 tonnes at 1,600 tonnes per day was from an open pit. This ore was crushed to < 13 mm and transferred to leach pads where the gold was recovered by heap leaching. Waste rock was used as fill for construction and road building, with the excess sent to a waste-rock pile.

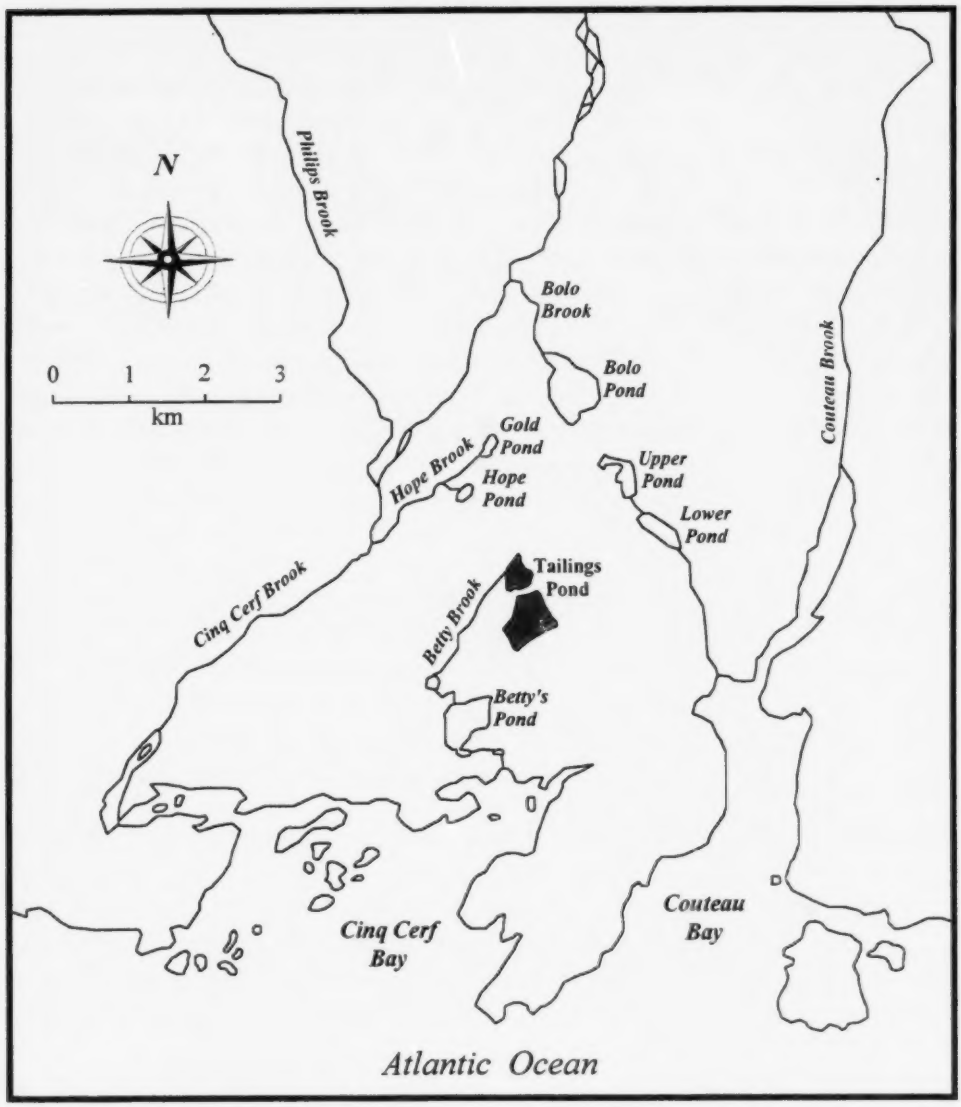
Full underground production commenced in 1989 at a rate of 3,000 tonnes per day. After crushing and grinding, coarse gold was recovered by gravity separation. Lime and cyanide were mixed with the ore, the slurry thickened, and the gold-cyanide solution sent to carbon columns for gold recovery. The solids were treated with more lime and cyanide in leach tanks, and further gold-cyanide solution extracted. Carbon was then added to the pulp to recover additional gold. The pulp was filtered to recover the cyanide solution, and the solids treated to remove cyanide and sized. Coarse material was used as backfill underground, and fines were discharged to the tailings pond.

Cyanide in the tailings was destroyed by the "Degussa process" which employed hydrogen peroxide, sulphuric acid, ferric sulphate, and TMT 15 (a proprietary organic reagent). This process did not work reliably or economically, and an alternative INCO SO_2 -air system was tried on a small scale.

Figure 3.1: Location of the Hope Brook Gold Mine



Fgure 3.2: Surface water drainage around the Hope Brook Gold Mine



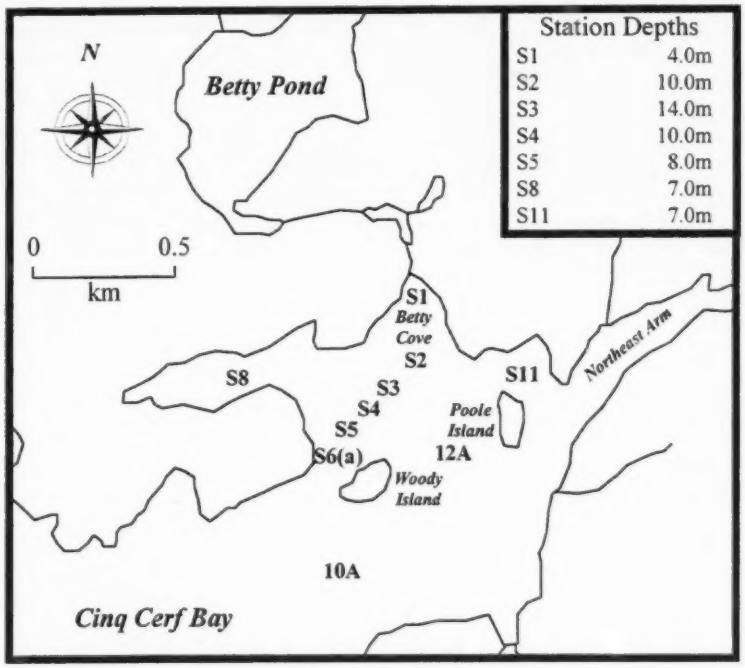
Because of economic and environmental problems, mining operations were suspended in May 1991, and the underground mine workings allowed to flood. In the one year period during the suspension of the operation the water in the primary tailing pond had been treated to reduce cyanide and copper concentrations. Treated water was released through the polishing pond into the Betty's Pond watershed and into the receiving marine bay (Cinq Cerf Bay which includes the Betty's Cove).

Royal Oak Mines Inc. acquired the rights to the Hope Brook Mine and re-opened the operation on July 1, 1992 (after recommissioning the mine/mill equipment and dewatering the mine workings), mining 105,000 tonnes per month from the underground operation. Reserves at December 31, 1993 were put at 11,251,307 tonnes containing 3.3 g/t gold. The mill capacity was increased to 3,500 tons per day, and the processing operations were modified slightly to increase gold recovery. Part of these modifications included a copper flotation circuit which has substantially decreased copper levels in the mill effluent. The other important change from an environmental point of view has been the change to the INCO SO_2-air system for cyanide treatment, which has significantly and consistently lowered cyanide effluent levels. A new effluent treatment system was implemented after re-opening the mine. The treated effluent stream is discharged into a primary tailings pond designed to allow the solid tailings to settle out by gravity. Water from the primary tailings pond is decanted into a secondary tailings pond referred to as Pine Pond. Pine Pond provides retention time in the order of 200 days which enhances further reduction in the concentrations of contaminants in the effluent by means of dilution by natural runoff, photodegradation of cyanides, and settling of colloidal metal precipitates. Water from Pine Pond is pumped into a small polishing pond which provides three days of water retention. Water from the polishing pond is decanted into Betty's Pond watershed which drains into the Cinq Cerf Bay. Department of Fisheries and Oceans in St. John's provided advice on procedures for the monitoring in 1992 and 1993.

3.2 Assessment of Monitoring Programs

The reports contain data from seven years (1987 to 1993) of monitoring of metals, particularly copper, zinc, lead, mercury, and iron, and, in some cases cadmium, cyanide, and ammonium (not frequently monitored) in water, sediments, brook trout, lobsters, and horse mussels. One report contains information on changes of benthic community in selected streams in the mine operation watershed. The location of the monitoring stations is illustrated in Figure 3.3.

Figure 3.3: Location of monitoring stations



3.2.1 Water Quality

Report Nos. 342, 345, 349, 616, 617, 674 cover the water quality sampling surveys conducted by the mine between the years 1989 and 1993. Since the surveys were conducted after the initial discharge of the mine, there is a lack of true pre-operational baseline data. The subsequent post-operational data, however, is still deemed to be adequate for interpretative purposes of temporal trends for future monitoring, and especially useful regarding spatial trends.

1989:

Water was sampled from five stations located in Betty's Cove. Stations were located at 25, 50, 100, 200, and approximately 500 m on a transect south from the outflow of the brook draining Betty's Pond. At each station, water was sampled from four depths: 0.5 m, 1 m, 3 m, and 6 m above the bottom. Samples were collected by 5-L Niskin bottle. Each sample was divided into three subsamples: one for metal determination; one for CN determination (subsample preserved with NaOH); and one for salinity and total suspended solids determination. The concentrations of cadmium, copper, lead, and zinc were determined by

Graphite Furnace Atomic Absorption Spectrometry. Total and weak acid dissociable cyanides, salinity, and total suspended solids were determined by colorimetry, ion chromatography, argentometric titrations, and filtration, respectively.

Concentrations of up to 470 µg/L copper were found in 0.5 m of water at the 100 m distance from the mouth of the brook draining Betty's Pond into Cinq Cerf Bay as compared to 3 µg/L copper at the 500 m distance from the brook mouth.

The concentration of copper in water in Cinq Cerf Bay near the mouth of the brook draining Betty's Pond exceeded the 0.3 mg/L limit given in the 1980 Newfoundland Regulation 156/80 for discharge of effluent into public sewer or into a water body. Copper was the only element in water reported by the consultants who carried out the monitoring. Total cyanide up to 1.2 mg/L was detected in surface water at the 100 m distance from the brook's mouth as compared to < 0.005 mg/L cyanide in surface water at the 500 m distance from the brook's mouth.

1992:

Water samples were collected at each of the 1992 sediment sampling stations in Betty's Pond at 1 m below water surface and 1 m above the bottom. Surface water samples were collected at each of the 1992 sediment sampling stations in Cinq Cerf Bay. The determination of total and soluble copper, total cyanide, total zinc, total iron, ammonium, and pH in Betty's Pond samples and of total copper, total cyanide, and ammonium were carried out at the Hope Brook assay laboratory.

Concentrations of all parameters were lower than those given in the 1980 Newfoundland Regulation 156/80 for discharge of effluent into public sewer or into a water body.

1993:

Water samples were collected at 10 stations in Cinq Cerf Bay (locations similar to 1992), and at three stations in Betty's Pond. Water was collected at surface (1 m below water surface), middle of the water column and 1 m above the bottom.

The only obvious incident of non-compliance respecting Provincial or Federal Statutes was the exceedence of the 0.3 mg/L copper limit of the Newfoundland Regulation 156/80 in 1989 for discharge of effluent into a public sewer or into a water body. Concentrations of copper and other parameters covered by the Regulation were within accepted limits in subsequent sampling surveys. Mention of an increase in copper levels in Cinq Cerf Bay during the 1993 survey was attributed to analytical error.

The collective water sampling program conducted at the mine site is not deemed suitable to make any inference to "impact-no impact" to the receiving environment. The data, however, can provide a qualitative insight into trends and serve as the foundation for more rigorous future monitoring programs.

3.2.2 Sediment Quality

Report Nos. 342, 345, 349, 616, 617, 674 cover the sediment quality sampling surveys conducted by the mine between the years 1987 and 1993. No data set includes any indication of sufficient description of sampling/analytical methodologies and QA/QC to have complete confidence in the data. Also, where protocols are discussed, there are obvious concerns regarding the comparability of data, especially for temporal trends, due to changing methodologies and obvious analytical/sampling errors. Similar to the water quality data, caution should be exercised in making definitive conclusions on "impact-no impact". Nevertheless, there are clear indications of improvements in sampling methodologies (e.g., use of fine grain fractions for chemical analyses) in later sampling years.

1987-1988:
Three replicates of sediment samples were collected at five stations in Betty's Pond and five stations in Betty's Cove in 1987 and in 1988. Samples were collected using Ekman grab. Surficial 1 to 2 cm of sediments were subsampled.

1989:
The same sampling stations were used as in 1987/88 in Betty's Pond. Sediments were sampled at ten stations in Betty's Cove and Culotte Cove. Five replicates were taken at each station using an Ekman grab, subsampled surficial 1 to 2 cm of the sediments. At each station, the five replicates were made into one composite sample. Frozen samples were shipped to the lab. Metals were determined by Atomic Absorption Spectroscopy.

1990:
Sediments were collected in triplicate at 10 stations in Betty's Pond using an Ekman grab. Surface (1 to 2 cm) sediment was collected from each sample, and the triplicate samples were made into a composite sample. Sediments were collected at ten stations in Betty's Cove and Coulotte Cove. Some sampling locations from the previous year's monitoring were changed to provide information on the effects of the tailings treatment system implemented in the mine. Samples were collected by Ekman grab in five replicates at each station in Betty's Cove and Coulotte Cove. Surface (1 to 2 cm) sediment was subsampled from each

replicate and composited. Analytical methods were similar to those used in 1989 (Atomic Absorption Spectroscopy). QC/QA was included in the analysis.

1991:
Sediment samples were collected from 10 stations in Betty's Pond (same stations as in 1990) in triplicate by Ekman Grab. Sampling procedures were similar to those in 1990. Sediment samples were obtained from 10 stations in Betty's Cove, Culotte Cove and Northeast Arm of Cinq Cerf Bay, as in 1990. Sample preparation included sieving through a 1-mm mesh. Sediments were acid-digested (nitric, hydrofluoric, perchloric acids) in Teflon bombs prior to analysis by ICP/MS. Cyanide was determined on wet samples using colorimetry.

1992:
Sediments were collected from 10 stations in Betty's Pond. Station locations were the same as in previous years. Sediments were collected at 9 stations in Cinq Cerf Bay (location similar to previous years). Sediment cores were collected by hand from a small boat. Surface (2 cm) sections were obtained from the cores, air dried, and sieved through 1-mm mesh. Analytical methods were the same as in 1991.

1993:
Sediment samples were collected at the same locations in Betty's Pond and in Cinq Cerf Bay as in 1992. Samples were collected by divers using hand coring. Surficial 2 cm of the sediment cores were used in the analysis. Samples were screened through 230 mesh (63 μm opening) prior to the sample processing. The concentrations of cadmium, copper, lead, mercury, zinc, cyanide, particle size, and organic matter were determined in all samples. Analytical methods were similar to 1992.

There were no apparent incidents of non-compliance respecting Provincial or Federal Statutes (e.g., Federal Ocean Dumping Regulations for cadmium and mercury).

The collective sediment sampling program conducted at the mine site is not deemed suitable to make any inference to "impact-no impact" to the receiving environment. The data, however, can provide a qualitative insight into trends and serve as the foundation for more rigorous future monitoring programs.

3.2.3 Fish - Tissue Residues

Report Nos. 342, 345, 349, 616, 617, and 674 cover the fish contaminant sampling surveys conducted by the mine between the years 1987 and 1988. No data set includes any indication of sufficient description of sampling/analytical methodologies and QA/QC to have complete confidence in the data. Also, fishing effort to obtain samples in 1989 was unsuccessful and the program was discontinued in 1990. (It was indicated that the species no longer inhabits Betty's Pond, which may suggest impacts to the species attributable to mining activities.) Similar to the water quality and sediment data, caution should be exercised in making definitive conclusions on "impact-no impact".

1987-1988:
Brook trout were collected in 1987 in Northwest Pond (control site) and in 1987 and 1988 in Betty's Pond. Methods for sampling, sample handling and analysis are described only in a proposed protocol for sampling, etc. The report does not mention if recommended procedures were followed. Proposed protocol (the only available information on sampling and analysis in the report) recommended to collect 30 specimens of 3+ age class and 5 specimens for each 1+, 2+, 4+ and 5+ age classes; record the length, weight, sex/stage of maturity; gut fish and ship to the laboratory frozen in plastic bags. Analytical methods similar to those proposed for lobster tissue.

The collective fish residue sampling program conducted at the mine site is not deemed suitable to make any inference to "impact-no impact" to the receiving environment. The data, however, can provide a qualitative insight into trends and serve as the foundation for more rigorous future monitoring programs.

3.2.4 Invertebrates - Tissue Residues

Report Nos. 342, 345, 349, 616, 617, 674 cover the lobster and mussel contaminant sampling surveys conducted by the mine between the years 1987 and 1993. The majority of data sets include an indication of sufficient description of sampling/analytical methodologies and QA/QC. Half of the data sets, however, indicate varying degrees of analytical/sampling problems.

1987-1988:
Ten lobsters were collected at each Betty's Cove site and the control site at Captains Island. In 1987 and 1988 analyses were carried out on claw, tail and digestive gland. Methods for sampling, sample handling and analysis are described only in a proposed protocol for

sampling, etc. The report does not mention if recommended procedures were followed. The proposal recommended to collect 15 specimens of 1 to 2 pounds from each of Betty's Cove and the control site sampling stations. It was recommended that the lobsters be supplied by local fishermen, and stored and shipped alive to the lab in a sealed cooler. It was recommended to homogenize the tissue with subsequent air drying, acid-digest the dry tissue and analyze for metals using ICP or flameless atomic absorption spectroscopy. No significant difference in metal concentrations were found between stations in 1987 and 1988. This suggests that the control site was not properly selected.

Mussels were collected at 4 stations in Betty's Cove and one control site east of Captains Island in 1987. In 1988, there was an insufficient number of specimens; location of the stations, therefore, was changed from 1987. Methods for sampling, sample handling and analysis were described only in a proposed protocol for sampling, etc. The report does not mention if recommended procedures were followed. However, recommended sampling, sample processing, and analysis were similar to those recommended in lobster monitoring. Data obtained for the other metals are difficult to compare because of the change in location of the sampling stations from 1987 to 1988.

1989:

Commercial fishermen supplied lobsters from the control sites (Couteau Bay and Captains Island) and from the Betty's Cove area. Twenty specimens were used in the analysis. The hepatopancreas was separated, homogenized, and subsampled for determination of mercury and other metals (as in previous years). The tissue was digested by nitric acid. Metals were determined by flame atomic absorption spectroscopy and graphite furnace atomic absorption spectroscopy. The samples digested by sulphuric and nitric acids (procedure described by Water Quality Branch, IWD, 1979) to determine mercury.

Mussel samples were collected at 4 sites in Betty's Cove and at one control site (east of Captains Island). The mussels were depurated for 24 hours in uncirculated water. The analytical methods were similar to those used in analysis of lobster tissue (ICP/MS).

1990:

Ten lobsters were collected from Betty's Cove and control sites at Captains Island (10 specimens) and Barasway Island (10 specimens) by local fishermen. A problem with labelling the lobsters devalued all of the obtained data.

Fifteen to twenty mussels were collected from each of the four 1989 stations in the subtidal zone (1 to 6 m) in Betty's Cove by scuba diving. The collected mussels were depurated for 24 hours in uncirculated water. Tissue from 10 selected specimens from each station was

homogenized and analyzed. The analytical methods used were the same as in 1989. ANOVA was used in the statistical evaluation of the data. Some sampling stations established in 1987-1988 had to be moved or cancelled because of the lack of mussels.

1991:
Sample collection from Betty's Cove and Barasway Island (control site) was similar to 1990 (samples were correctly labelling in 1991). Lobster tail muscle was sampled in 1991 to compare data from 1987 to 1991. Lobster hepatopancreas tissue and tail muscle was analyzed for cadmium, copper, mercury, lead, and zinc using ICP/MS. The moisture content in lobster tissue was determined to allow the results to be expressed on wet and dry weight basis.

Sampling methods were similar to those used in 1990. The moisture content in the mussel tissue was determined to allow the results to be expressed on wet and dry weight basis. All data was reported on wet weight basis. Determination of metals was by ICP/MS.

1992:
Seven lobsters were obtained from each of two sampling stations (one in Cinq Cerf Bay east of Woody Island; one from a control station at Barasway Island). The lobsters were dead upon arrival in the lab (ascribed to overheating in transit). The analytical methods were similar to those used 1991. QA/QC was included in the analysis. Lobster hepatopancreas was analyzed. There were problems encountered in getting a sufficient number of samples.

Mussels were collected by a diver from 6 stations (five in Betty's Cove, one from a control station at Captains Island). The mussels were purified for 72 hours in uncirculated water. The samples were dissected and dried at 55° C. The analytical methods were similar to those used in 1991.

1993:
Ten lobsters were obtained from two sampling points: one in Cinq Cerf Bay and one at the control site at Barasway Island. The analytical methods were similar to those used in 1992.

The sampling and analytical methods for mussels were similar to those used in 1992. In the description of sampling methods in 1993, there was no mention of purification of collected mussels. There were some problems with the thawing of frozen samples during shipping to the lab.

The collective invertebrate residue sampling program conducted at the mine site is not deemed suitable to make any inference to "impact-no impact" to the receiving environment.

The data, however, can provide a qualitative insight into trends and serve as the foundation for more rigorous future monitoring programs.

3.2.5 Invertebrates - Population Assessment

Report Nos. 342, 345, 349, 616, 617, 674 cover the invertebrate population sampling surveys conducted by the mine between the years 1986 and 1989. No data set includes any indication of sufficient description of sampling/analytical methodologies and QA/QC to have complete confidence in the data.

1986-1989:
In September 1985, triplicate Surber samples were collected at several sites within the Cinq Cerf Brook watershed. These collections were part of the baseline data required for the Aquatic Environment Component Study of the Hope Brook Gold Mine EIS. From 1986 to 1989, triplicate Surber sample collections were taken each June/early July at two sites in Cinq Cerf Brook and one site in Hope Brook. Two of these sites were sampled in 1985. Animals identified to lowest taxonomic levels possible.

The collective invertebrate population sampling program conducted at the mine site is not deemed suitable to make any inference to "impact-no impact" to the receiving environment. The data, however, can provide a qualitative insight into trends and serve as the foundation for more rigorous future monitoring programs.

3.3 Results of Aquatic Effects Assessment

3.3.1 Water Quality

According to the 1989 surface water survey, there appears to be a sharp gradient-to-background effect for copper (up to 470 µg/L copper in surface water at the 100 m distance from the mouth of the brook draining Betty's Pond into Cinq Cerf Bay as compared to 3 µg/L copper at the 500 m distance from the brook's mouth). Also, similar trends are noted for cyanide (total cyanide up to 1.2 mg/L in surface water at the 100 m distance from the brook's mouth as compared to < 0.005 mg/L cyanide in surface water at the 500 m distance).

3.3.2 Sediment Quality

There is a general temporal trend of increasing sediment metals levels over the 5 year sampling period. There are increased concentrations of copper, lead, zinc, and iron in Betty's Cove sediments from 1987 to 1988. Mean concentrations increased from 1987 in Betty's Pond sediments up to about: cadmium 2x; copper 71x; ,mercury 300x; and lead 7.6x. Mean concentrations increased from 1987 in Betty's Cove sediments up to about: cadmium 6x; copper 4x; lead 9.7x; and zinc 2x.

The concentrations of copper in Betty's Pond sediments increased about 6.5x (up to 2,866 μg/g copper) from 1989. Considerable increase of all metals (cadmium, copper, mercury, lead, zinc) in Betty's Pond sediments.

In Betty's Cove average concentrations of cadmium, copper, mercury, lead, and zinc increased from 1987 (cadmium about 2x; copper about 2.5x; mercury about 16x; lead about 5x; and zinc about 1.2x above the 1987 values). Mean concentrations of cadmium, copper, and cyanide in sediments increased significantly from 1990 (about 3x, 5.3x, and 1.9, respectively). Concentrations of copper, mercury, cadmium, lead, and cyanide in sediments decreased from those in 1991 in both Betty's Pond and Betty's Cove. The concentrations of cadmium, copper, lead, mercury, zinc, and cyanide in sediments from Betty's Pond and Cinq Cerf Bay increased from 1992.

Sieving of sediments through different openings in 1992 and 1993 most likely affected the determination of concentrations of metals and cyanide in the sediments. In 1993, only fine-grained sediments (<63μm) were used in analysis. These usually contain greater concentrations of contaminants than the coarse sediments.

3.3.3 Fish - Tissue Residues

All age classes of collected brook trout contained similar concentrations of metals. No differences in concentrations of metals in brook trout samples collected in 1987 and 1988. Even though there are no apparent temporal trends regarding metal uptake in brook trout (*S. fontinalis*), the fact that all age classes of fish contained similar concentrations and that only one sampling year was completed which generated results, it is unclear whether spatial or temporal trend do not exist.

3.3.4 Invertebrates - Tissue Residues

There appears to be an increase in invertebrate tissue residues for metals between 1987 and 1993. For lobster, the concentrations of copper, zinc, lead, and iron were elevated in samples collected in 1988 compared to 1987 (elevation statistically significant, actual values of concentrations confusing). Increase of copper in lobster hepatopancreas from 1987: up to 22.8x, i.e., increase of mean concentrations of copper in samples collected in Betty's Cove from 1987 to 1989. Increase in mean concentrations of copper in lobster hepatopancreas in Betty's cove from 1990 (about 3.3x) and in zinc (about 1.2x). Increase in copper (about 3.9x), mercury (1.1x), and zinc (1.2x) and decrease in cadmium (1.2x) in lobster hepatopancreas from 1991. High variability in obtained data confused a meaningful interpretation. The data in the report, however, indicated an increase of mean concentrations of zinc and copper in lobster hepatopancreas samples from Cinq Cerf Bay from 1992 to 1993.

For horse mussels, changes were less discernible. The concentration of lead in mussels increased from 1987 to 1988 at all sampling sites, including the control. There was an increase of metal concentrations in mussel tissue from 1987 up to about: cadmium 71x; copper 9.6x; mercury 88x; lead 4.2x; zinc 2.8x. The concentrations of metals similar to those in 1989 (no significant increase). Generally, similar concentrations of metals were observed in mussel tissue as in the 1990 monitoring (some decrease in mercury, lead, and zinc concentrations from 1990). The concentrations of cadmium, mercury, lead, and zinc in mussel tissue at the control station increased between 1987 and 1989. The concentrations of copper remained similar over the monitoring period (1987 to 1992). No explanation given in the report(s). Sampling methods were similar to those used in 1992. Analytical methods were similar to those used in 1992. In the description of sampling methods, there was no mention of purification of collected mussels in 1993.

3.3.5 Invertebrates - Population Assessment

No significant changes were observed in the benthic community structure at the sampling sites during 1986 to 1989.

3.4 Discussion

Most of the data obtained for this case study was generated from successive surveys, with the intention to provide good temporal and spatial trend information on environmental quality at this particular site. Unfortunately, as with many of the other case studies, this information is sparse and the interpretation is hampered because of various reasons (e.g.,

inconsistent use of methodologies, insufficient QA/QC, natural variability, lack of adequate pre-operational baseline data, etc.). In general, the chemical and biological monitoring methodologies were useful in addressing environmental quality issues in the vicinity of the mine. With the proper implementation of QA/QC practices and proper study design, the value of such monitoring programs could be greatly enhanced.

3.5 Conclusions

1. Although extensive environmental monitoring has been conducted at this mining operation, the program has been inconclusive because of inconsistent methods and approaches.

2. Interpretation of the results of the monitoring program at the Hope Brook mine have been complicated by lack of rigid quality assurance.

3.6 Recommendations

1. In order to assess trends in environmental quality, monitoring programs should follow a consistent approach so that year-to-year comparison of results can be made.

2. All environmental monitoring programs should adhere to a sound quality control and quality assurance program.

3.7 References

References from AQUAMIN database, by report number

342 Hope Brook Gold Inc. Interpretation of the metals monitoring program, 1987, 1991. (1991): Environment Protection Laboratories, Inc., Toronto, Ont.

345 Hope Brook Gold Mine heavy metals monitoring program: 1990 sampling, analysis 1987-1990. LeDrew, Fudge and Associates Limited (1991): St. John's, NF.

349 Hope Brook Gold Mine heavy metals monitoring program. LeDrew, Fudge and Associates Limited (1991): St. John's, NF.

616 Hope Brook Gold Inc.: Heavy Metals Monitoring Program Report - 1988. BP Canada, Inc. (1989): Prepared for Hope Brook Gold Inc. 17p. + appendices.

617 Hope Brook Gold Mine: Heavy Metals Monitoring Program, 1989 Sampling. Ledrew, Fudge Associates Ltd. (1990): Prepared for Hope Brook Gold Inc., Corner Brook, Nfld. 21p. + appendices.

674 Royal Oak Mines, Inc., Newfoundland Division, Hope Brook Mine, 1993 Environmental Effects Monitoring Program (1993)

676 A Summary Report of the Annual Monitoring of Benthic Stream Invertebrates in Cinq Cerf Brook and Hope Brook, 1985-89. LeDrew, Fudge and Associates, Ltd. (1990)

678 Royal Oak Mines, Inc., Newfoundland Division, Hope Brook Mine, 1992 Environmental Effects Monitoring Program. (1992)